RANGELANDS AT EQUILIBRIUM AND NON-EQUILIBRIUM

RECENT DEVELOPMENTS IN THE DEBATE AROUND RANGELAND ECOLOGY AND MANAGEMENT

EDITED BY SUSANNE VETTER

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papers from the international workshop on 'rangelands at equilibrium and
non-equilibrium' held at the VIIth international rangelands congress
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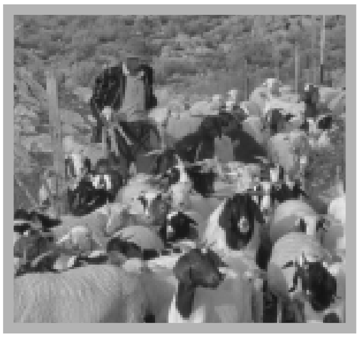

INTRODUCTION

Susanne Vetter

BACKGROUND

The debate on equilibrium vs non-equilibrium dynamics in pastoral systems emerged in the early 1980s, when economists, ecologists and social scientists began to challenge the widespread claims of overgrazing and degradation in African rangelands and subsequent interventions based on rangeland succession theory and correct stocking rates (for example, Sandford 1982; 1983; Homewood & Rodgers 1987; Ellis & Swift 1988; Abel & Blaikie 1989; Westoby et al. 1989). The debate gained momentum in the early 1990s after two international workshops around emergent new paradigms in rangeland ecology and socio-economics (Woburn I and II), which resulted in the publication of two books, *Range Ecology at Disequilibrium* edited by Behnke et al. (1993) and *Living with Uncertainty* edited by Scoones (1994). The 'new rangeland ecology' argued that the traditional, equilibrium-based rangeland models did not take into account the considerable spatial heterogeneity and climatic variability of semi-arid rangelands, and that mobility, variable stocking rates and adaptive management were essential for the effective and sustainable utilisation of semi-arid and arid rangelands.

Possibly the most widely cited and debated argument of the non-equilibrium rangeland ecology was that plant composition and biomass in semi-arid rangelands are primarily driven by rainfall and not by grazing pressure, that animal numbers are kept below equilibrium densities by frequent droughts, and degradation of the vegetation as a result of overgrazing is thus unlikely (Ellis & Swift 1988; Behnke & Scoones 1993). Ironically, while one of the aims of the 'new rangeland science' was to highlight the complexity of rangelands that had been ignored and suppressed by the old 'mainstream' paradigm, much of the debate seems to have centered around a highly simplified dichotomy of 'equilibrium and vulnerable to degradation' and 'non-equilibrium and resilient'. This argument became the focus of a heated debate, which was fuelled by concern that the idea: 'semi-arid rangelands don't degrade' would be as uncritically adopted in rangeland management and policy as previously the equilibrium-based rangeland succession model had been.

At the same time, the debate has stimulated much research and some new dialogue among researchers from different disciplines. The scope of research has expanded to include ecological models more complex than the equilibrium–non-equilibrium dichotomy: the dynamics of pastoral areas beyond tropical Africa, a recognition of the various temporal and spatial scales that need to be explicitly taken into consideration, and explorations of the power dynamics at play in applying rangeland science to pastoral systems. This has coincided with a move in pastoral development and policy towards more participatory approaches. Despite these various trends and activities in the research community, however, equilibrium ideas of predictability and stability appear to remain entrenched in most agricultural departments and development organisations.

One of the most striking features of the debate is its multi-disciplinary nature. In recent years, researchers and practitioners from different backgrounds have made an increased effort to communicate across disciplines, and to engage with the arguments presented by researchers in different fields. This collection of papers arose from a workshop held at the VIIth International Rangelands Congress in Durban, South Africa, in July 2003. The aim of the workshop was to bring together an inter-disciplinary group of researchers and practitioners to take stock, ten years after the Woburn workshops, of where the debate had moved, what research has been done to test the alternative models, and to identify directions for a future research agenda. Participants at the workshop included several of the original Woburn participants, as well as researchers who have become interested in non-equilibrium more recently and people in the policy and development arena. The papers were prepared to review developments in the debate and to stimulate discussion at the workshop. Together, they illustrate the diversity of approaches and also – inadvertently – some of the challenges encountered in the dialogue between the disciplines with their different languages, research approaches and ideologies.

SUMMARY OF PAPERS

The first paper by Susanne Vetter introduces the debate. It outlines the predictions and management implications of the equilibrium and non-equilibrium paradigms, reviews some recent research to test their predictions, and discusses the context in which the ecological debate is situated.

Andrew Illius and Tim O'Connor outline an ecological definition of non-equilibrium and the role of key resources in maintaining livestock populations. Their contribution is a brief synopsis of arguments presented in more detail in two of their papers (Illius & O'Connor 1999; 2000).

David Richardson, Brian Hahn and Timm Hoffman present two long-term simulation models to examine the applicability of equilibrium and non-equilibrium theory to semi-arid rangelands. Their models were developed for semi-arid, winter rainfall shrublands in South Africa and show that rainfall is the overriding driver of livestock populations. Livestock do, however, affect vegetation composition via selective utilisation of different plant guilds. The authors conclude that neither equilibrium nor non-equilibrium models are adequate on their own to predict plant or animal dynamics. Furthermore, they indicate that the generalisability of simulation models across sites with different vegetation is very limited.

Pierre Hiernaux reviews equilibrium and non-equilibrium concepts as they apply to vegetation communities of the Sahel. His paper highlights the interaction of abiotic factors (such as rainfall, fire and floods) and grazing in shaping vegetation communities. He concludes that grazing can have an impact on vegetation even in fairly arid, largely non-equilibrium rangelands, but that the effect of this depends on the timing and duration of grazing relative to the growing season, as well as the species composition of livestock herds. Appropriate management of such systems must take this into account, and should be based on mobility and adaptive management rather than simply the maintenance of recommended average stocking rates.

David Ward uses results from studies in Namibia to discuss the relationship between communal grazing and degradation in arid areas. The paper shows that degradation is not necessarily an outcome of high human and livestock densities in arid areas over time scales of a century or less. Comparisons between commercial and communal livestock farming areas highlighted that stocking rates per se are not the most important determinants of plant cover, productivity, biodiversity and soil parameters (which did not differ significantly between the two systems). The boom and bust pattern of communal livestock populations and the lower but constant stocking rates in commercial areas (which result in higher post-drought grazing pressure) may in fact have similar negative effects on the vegetation over long time scales. The paper concludes by highlighting the importance of considering social and historical factors when assessing change and degradation in pastoral systems. Often, factors external to communal pastoralism – such as national policies, climate change, or construction of dams upstream, which reduce production potential, occur at larger scales and are out of the control of the land users.

Carol Kerven explores the applicability of non-equilibrium concepts, which so far have been largely applied and debated in an African context, to the cold rangelands in northern Asia. Her paper explores parallels and differences between northern Asia and the drylands of Africa. Like drought in Africa, extreme cold events in northern Asian rangelands regularly decimate livestock populations in a density-independent manner. Pastoralist strategies traditionally emphasise mobility (with long-distance transhumance), flexibility and social networks that act as buffers to environmental uncertainty. As in Africa, political and economic interventions in Asia have had a profound impact on the ways in which pastoralists have coped with climatic variability. The Soviet system restricted mobility and instead prevented winter mortality by producing livestock fed on a massive scale. This reduction of mobility and the maintenance of high livestock numbers has resulted in widespread rangeland degradation. Since state support for livestock farmers has dwindled with the change-over to a market economy, pastoralists are again vulnerable to winter extremes, without the ability to revert to their former strategy of transhumance for a variety of reasons.

Stephen Sandford reviews some issues in the economic assessment of livestock systems in Africa, particularly the comparison of the relative benefits of conservative and opportunistic stocking rates. He explores different factors that affect the outcomes of economic models, such as the types and proportions of benefits and commodities measured, the economic criteria used to assess their value, the exact sequence of climatic events and when in this sequence one starts modelling. A comparison of different models reveals that their outcomes under the same conditions can differ vastly. Given the lack of generalisability of models across sites, the sensitivity of model outcomes to certain assumptions and parameter values, and the relatively high complexity of even 'simplified' models, the usefulness of simulation models as a decision-making tool for policy and decision-making in Africa is debateable. Yet there is no escaping the fact that the choice between conservatism and opportunism has important economic and environmental consequences. Given the complexity of the systems under study, the vast number of possible climatic and economic scenarios and the long time-frames of analysis, using simulation models is the only feasible option for comparing the outcomes of different economic strategies. Their results must be interpreted with caution, however, and blanket statements about the superiority of

one or the other strategy should be viewed with scepticism.

Wolfgang Bayer and Ann Waters-Bayer discuss the difficulties of translating non-equilibrium concepts into pastoral development practice. Although it has long been recognised that climatic variability and regular droughts are a reality in pastoral areas, and that a flexible, adaptive approach to management is appropriate, little has changed in the fundamental approach of development programmes. The authors propose that this is not merely a result of a lack of scientific evidence, but that power relations – between donors, development agencies, governments of the recipient countries and the pastoralists – shape which 'truth' determines action. Because money determines power, it is not surprising that pastoralists have the least say in determining the research and development agenda.

The final chapter summarises common themes, unresolved debates and future research challenges that emerged from workshop discussions. It has become clear that temporal variability and spatial heterogeneity are a reality in many of the world's rangelands (not only in semi-arid tropical Africa). Ecologically speaking, both equilibrium and non-equilibrium dynamics are found in rangelands, often at different times or governing different parts of the resource. The spatial complexity and year-to-year unpredictability of these systems presents a particular challenge to researchers, agriculturalists and policy makers, not to mention the land users themselves. In addition to the uncertainty imposed by the ecological system, external pressures and interventions are having an impact on pastoralist societies and on the natural resources they rely on. Pastoral strategies to cope with the uncertainty of their environment centre around mobility, flexibility, opportunism and access to an expanded resource base – in the form of land, employment and kinship networks – during droughts and other threats. Fragmentation and constriction of pastoral landscape through fencing, privatisation and transformation to other land uses (mainly conservation or cropping) has led to sedentarisation and concentration of pastoral activities. This has resulted in rangeland degradation in many places. While researchers now appreciate much of this, the challenge remains to translate this knowledge into recommendations, policy and management practice.

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REFERENCES

Abel, N & Blaikie, P. 1989. Land degradation, stocking rates and conservation policies in the communal rangelands of Botswana and Zimbabwe. *Land Degradation and Rehabilitation,* 1:101–23.

Behnke, R & Scoones, I. 1993. Rethinking range ecology: Implications for range management in Africa, in *Range Ecology at Disequilibrium: New models of natural variability and pastoral adaptation in African savannas*, edited by R Behnke, I Scoones & C Kerven, London: Overseas Development Institute.

Behnke, R, Scoones, I & Kerven, C. (eds.). 1993. *Range Ecology at Disequilibrium: New models of natural variability and pastoral adaptation in African savannas*. London: Overseas Development Institute.

Ellis, J & Swift, D. 1988. Stability of African pastoral ecosystems: Alternate paradigms and implications for development. *Journal of Range Management,* 41:450–9.

Homewood, K & Rodgers, W. 1987. Pastoralism, conservation and the overgrazing controversy, in *Conservation in Africa: People, policies and practice*, edited by D Anderson & R Grove. Cambridge: Cambridge University Press.

Illius, A & O'Connor, T. 1999. On the relevance of non-equilibrium concepts to semi-arid grazing systems. *Ecological Applications,* 9:798–813.

Illius, A & O'Connor, T. 2000. Resource heterogeneity and ungulate population dynamics. *Oikos,* 89:283–94.

Sandford, S. 1982. Pastoral strategies and desertification: Opportunism and conservatism in dry lands, in *Desertification and development: Dryland ecology in social perspective*, edited by B Spooner & H Mann. London: Academic Press.

Sandford, S. 1983. *Management of Pastoral Development in the Third World.* New York: John Wiley and Sons.

Scoones, I. 1994 (ed.). *Living with Uncertainty: New directions in pastoral development in Africa.* London: Intermediate Technology.

Westoby, M, Walker, B & Noy-Meir, I. 1989. Opportunistic management for rangelands not at equilibrium. *Journal of Range Management,* 42:266–74.

EQUILIBRIUM AND NON-EQUILIBRIUM IN RANGELANDS – A REVIEW OF THE DEBATE

SUSANNE VETTER

INTRODUCTION

For most of this century, there has been concern about the sustainability of communally grazed rangelands in Africa and other parts of the world. Pastoral systems are commonly viewed as overstocked, overgrazed, degraded and unproductive (for example, Lamprey 1983), and this has resulted in widespread interventions to reduce stock numbers in an attempt to halt degradation. Overgrazing is thought to be inevitable in communal pastoral systems because people keep more livestock than they need for a variety of reasons (for example, Lamprey 1983), and because of the problems inherent in communal ownership of the resource, where individual benefit is maximised at the expense of the communal resource (Hardin 1968). Increasing human population pressure, encroachment of rangelands by other land use, control of livestock diseases and the breakdown of traditional resource management structures are thought to contribute to the degradation problem.

This way of viewing and managing communal grazing systems has come under considerable criticism regarding its underlying ecological and economic assumptions, and the idea that communal rangelands are necessarily mismanaged is now widely challenged (for example, Sandford 1983; Homewood & Rodgers 1987; Ellis & Swift 1988; Abel & Blaikie 1989; Behnke & Scoones 1993; Behnke & Abel 1996; Sullivan 1996; Sullivan & Rohde 2002). From a broader debate about interpretations of desertification, and the identification of pastoralists as its major causative agent, a debate around the ecological dynamics and appropriate management of semi-arid rangelands has developed (for example, Homewood & Rodgers 1987; Leach & Mearns 1987; Dodd 1994). This debate arose in the 1980s in response to a growing concern that interventions aimed at stabilising spatially and temporally variable rangelands were inappropriate and damaging to pastoral livelihoods (Sandford 1983; Ellis & Swift 1988). At the same time, there was an increasing recognition in ecology outside rangeland systems that equilibrium dynamics were difficult or impossible to demonstrate conclusively in many ecological systems (Wiens 1977; 1984; 1989; Connell & Sousa 1983; DeAngelis & Waterhouse 1987), and that a different paradigm was necessary to describe non-equilibrium systems. Ellis & Swift (1988) and Westoby et al. (1989) applied non-equilibrium concepts to rangeland systems and pointed out that a fundamental misunderstanding of their ecological dynamics was leading to inappropriate and failed interventions.

Central to this debate is the relative importance of biotic and abiotic factors in driving primary and secondary production in rangelands, and the consequences of this regarding the potential for grazing-induced rangeland degradation. The equilibrium model stresses the importance of biotic feedbacks between herbivores and their resource and with each other. These are manifested in the form of density-dependent regulation of livestock populations, and the feedback of livestock density on vegetation composition, cover and productivity. Ideas of carrying capacity, optimal stocking rate and range condition are central to range management under this model. When livestock numbers exceed the carrying capacity of the environment, overgrazing leads to degradation via undesirable vegetation change and loss of cover (for example, Heady & Child 1994; Tainton 1999).

In contrast, it is thought that non-equilibrium rangeland systems are driven primarily by stochastic abiotic factors, notably variable rainfall, which results in highly variable and unpredictable primary production. The concept of carrying capacity is of little utility in non-equilibrium systems. Livestock populations are thought to have negligible feedback on the vegetation as their numbers rarely reach equilibrium with their fluctuating resource base. Degradation due to overgrazing is thus unlikely, as

vegetation composition, cover and productivity respond primarily to rainfall (Ellis & Swift 1988; Behnke & Scoones 1993).

The debate has stimulated much new research, and many researchers now agree that equilibrium and non-equilibrium are extremes along a continuum and that a range of conditions exists in between. However, the equilibrium ideas of stability and predictability have remained pervasive in ecology and range management and also in the social sciences (Scoones 1999). The challenge is now to understand under what circumstances different dynamics apply, since the two models have fundamentally different consequences for policy and management. Interventions based on the equilibrium model focus on reducing stocking rates and increasing stability, while the non-equilibrium paradigm advocates the facilitation of opportunistic stocking strategies and promoting mobility. The different predictions of the two paradigms also determine how 'safe' it is to allow livestock numbers to increase without causing degradation. Whether or not communal grazing is damaging to the environment in turn has profound and far-reaching consequences for the persistence of pastoral systems. For example, many pastoralist groups have been removed from their traditional grazing areas because they were seen as a threat to wildlife conservation in east Africa (for example, Homewood & Rodgers 1987). In southern Africa, the debate about degradation and low productivity in communal rangelands has influenced policy on land reform (Cousins 1996).

This paper introduces the predictions and management implications of the equilibrium and non-equilibrium paradigms, reviews some recent attempts to test their predictions and discusses the context in which the ecological debate is situated.

THE ECOLOGICAL DEBATE

WHAT DETERMINES THE SIZE AND PRODUCTIVITY OF LIVESTOCK POPULATIONS?

One of the ongoing debates in range ecology focuses on the relative importance of density dependent interactions and abiotic factors in determining herd productivity, reproduction and mortality from year to year (Ellis & Swift 1988; Illius & O'Connor 1999; Sullivan & Rohde 2002). In a grazing system with relatively predictable rainfall and hence forage production, livestock populations are regulated in a density-dependent manner via competition for food resources. As population size nears carrying capacity, increased competition for resources leads to reduced herd productivity and growth rate. A sign of density-dependence is that population growth rates decrease with increasing population size because of the effects of competition on reproductive and mortality rates. This is exemplified by the Jones and Sandland (1974) model of the effect of stocking rate on cattle weight gain.

Rainfall in different years affects grass production and composition (Dye & Spear 1982; O'Connor 1985; 1994; 1995) and hence the effective carrying capacity at different times. If livestock populations are near carrying capacity, and hence already competing for resources, they are likely to experience population crashes in drought years when resources become scarce (Caughley 1979). If livestock populations are well below the ecological carrying capacity, drought mortality is reduced because livestock are buffered against such stress events by greater forage reserves and better animal condition. This is considered to be a desirable state for a livestock farmer or pastoralists, and the recommended management practice is thus the maintenance of conservative stocking rates that are 'safe' in drier years.

In grazing systems with very high climatic variability, forage availability varies to such a great degree with rainfall that livestock population dynamics are driven by rainfall via its direct effect on forage availability in any given year, rather than by density-dependent interactions such as competition for resources (Wiens 1977). Livestock numbers build up during series of wet years. Mortality is high and density-independent during severe droughts, particularly droughts lasting longer than one year (Homewood & Lewis 1987; Ellis & Swift 1988; Scoones 1990; Oba 2001). Population size thus fluctuates dramatically, and cannot track rainfall closely because of the time it takes populations to recover from crashes.

The dichotomy between density-dependent and abiotically driven population dynamics is an oversimplification of the range of situations found in reality. The strength of density-dependent interactions varies over time and in space. For example, density-dependent dynamics in non-drought years can alternate with density-independent mortality during droughts and subsequent recovery (for example, Scoones 1990, in southern Zimbabwe, Desta & Coppock 2002, in southern Ethiopia). In a grazing experiment examining the relative importance of rainfall and stocking rate on plant composition, primary and secondary production, Fynn and O'Connor (2000) found that density-dependent consumer-resource coupling was largely limited to drought periods and greater at high stocking rates. This example illustrates that a measure of grazing pressure (number of livestock per unit of available forage) is more informative – if harder to quantify – than stocking rate (number of animals per area) in systems where livestock numbers and rainfall vary over time.

Illius and O'Connor (1999) contest that more arid, 'non-equilibrium systems' are qualitatively different in their underlying dynamics and argue that just because a system displays variability in rainfall and forage production, this does not make it an inherently non-equilibrial system.

Instead, livestock populations in arid and semi-arid grazing areas are regulated in a density-dependent manner by their key resources (that is, the dry season resource), while being largely uncoupled from the outlying wet season resource.

It is in fact exceedingly difficult to infer density-dependent mechanisms – or their absence – from population census data. Part of this problem lies in the extreme difficulty and cost involved in obtaining detailed enough data on population size, mortality, fecundity and migration over a long enough time series. Sometimes, different authors use the same livestock census data in different ways, depending on the variables they choose. Oba (2001), for instance, uses fluctuating livestock numbers over two drought and recovery periods to infer that population regulation is density-independent. However, when one plots herd growth rates during the post-drought recovery period against herd size (Figure 2, Oba 2001:383), a negative relationship emerges which can be interpreted as evidence that density-dependent regulation of population growth is taking place. More fundamentally, a focus on the phenomenon of density-dependence does not in itself provide an explanation of the underlying mechanisms of the consumer-resource dynamics, for example how they are affected by seasonal variability and spatial heterogeneity in forage quality and quantity (Owen-Smith 2002).

WHAT DETERMINES THE COMPOSITION AND PRODUCTIVITY OF VEGETATION COMMUNITIES?

The equilibrium paradigm is based on the assumption that every environment has a certain carrying capacity, which is determined by biophysical characteristics such as mean annual rainfall, soil type and other biophysical characteristics of the area, which determine the potential primary production (East 1984; Bell 1985; Fritz & Duncan 1994). The actual carrying capacity in an area at any given time is determined by range condition, a term referring to functional attributes of the vegetation such as its potential for forage production and resistance to soil erosion (Trollope 1990). Range condition is assessed as a function of grass composition, biomass and cover and is interpreted as a stage in plant succession that can be manipulated predictably with stocking rate (Dyksterhuis 1949; Foran et al. 1978; Trollope 1990). No, or very light, grazing allows the vegetation to reach its climax stage, whereas heavy grazing pushes it back to a pioneer stage dominated by generally low-quality grass and forb species typical of disturbed environments. In its simplest form, this response of the vegetation to grazing pressure is linear, reversible and predictable. Range management aims to maintain the vegetation in a maximally productive climax or subclimax stage, depending on vegetation type, by adjusting stocking rates, rotational grazing and resting, and applying appropriate burning regimes. Continuous intense grazing leads to vegetation changes such as the replacement of palatable grasses with less palatable plant species, replacement of perennial grasses by annuals, bush encroachment, lower standing biomass and reduced basal cover (for example, Kelly & Walker 1976; Coppock 1993; Ash et al. 1995; Fynn & O'Connor 2000). It is predicted that these in turn will result in a decrease in forage quality and availability, increased variability of primary production, accelerated soil erosion, and ultimately an irreversible decline in animal production unless stocking rates are reduced.

Rainfall is thought to affect the vegetation via a similar mechanism – drought reduces range condition by pushing the vegetation community towards a pioneer stage, while high rainfall has a beneficial effect on range condition. Rainfall and stocking rate interact, with low rainfall exacerbating the effects of high stocking rate, and high rainfall ameliorating it. There is thus recognition of rainfall variability in the equilibrium model, contrary to the common belief that this model assumes static climatic conditions. The fundamental difference between the predictions of the two models is that the equilibrium model sees drought as focusing the effects of herbivory (Illius & O'Connor 1999), whereas the non-equilibrium model sees drought as relieving the pressure of high stocking densities, by making grazeable vegetation unavailable (Sullivan & Rohde 2002), and by inducing density-independent livestock mortality which reduces grazing pressure (Ellis & Swift 1988; Behnke & Scoones 1993).

A recognition that the response of the vegetation to grazing, drought and fire is not always linear and reversible as predicted by the rangeland succession model led to the suggestion that thresholds exist between different rangeland states (Friedel 1991), and to the development of state-and-transition models as an alternative (Westoby et al. 1989). These are based on the observation that several alternative stable states exist, and that changes between states require certain sets of conditions. Changes from some states (for example, bush encroachment) to previous states can require major management inputs. Management of such systems should be opportunistic and create or take advantage of conditions that allow switches to a more desirable state. Although state-and-transition models are considered to characterise rangelands not at equilibrium (Westoby et al. 1989), they have been applied to rangelands in conjunction with succession-based models (Phelps & Bosch 2002).

In very variable systems, no such distinct states can be distinguished, and the main driving force determining forage availability and composition every year is stochastic abiotic factors such as rainfall. Due to the short duration of the growing season, the high frequency of droughts, and the great inter- and intra-annual

variation of rainfall in semi-arid rangelands, the available amount of forage fluctuates considerably between years. Livestock numbers are unable to track these sharp fluctuations, and the dynamics and productivity of the vegetation and livestock are thus uncoupled most of the time. Two-year droughts, which are accompanied by severe mortalities, also occur regularly. Herd size builds up gradually in wetter years following a drought, during which time the vegetation is relatively lightly grazed. In a system such as this, degradation due to overgrazing is highly unlikely, since animals seldom if ever reach densities at which they provide a negative feedback to the vegetation (Ellis & Swift 1988).

Research indicates that equilibrium and non-equilibrium are extremes along a continuum. Non-equilibrium dynamics predominate in more arid systems, while more humid systems display equilibrium characteristics, and many systems encompass elements of both (for example, Wiens 1984; 1989; Ellis et al. 1993; Ellis 1994; Stafford Smith 1996; Oba et al. 2000; Ward et al. 2000; Desta & Coppock 2002; Sullivan & Rohde 2002). Evidence from arid environments (for example, Ellis & Swift 1988; Ward et al. 1998; 2000; Sullivan 1998 cited in Sullivan & Rohde 2002; Fernandez-Gimenez & Allen-Diaz 1999) suggests that these systems are well described by the non-equilibrium paradigm. In their arid study areas, vegetation cover, composition and productivity were strongly determined by rainfall, while grazing intensity had a negligible influence. One of the reasons arid rangelands appear to be resilient to long-term intensive grazing is that the grass sward is dominated by annual grasses, which do not germinate or establish in the absence of rainfall. Grasses grow from a seed bank in subsequent wet years, with biomass production more or less proportional to the amount of rainfall. As Sullivan and Rohde (2002) argue, there may be literally no grass to overgraze in a drought year. However, Turner (1999) found that long-term grazing history does affect the composition and peak biomass production of annual grasslands in the Sahel, even though vegetation responses to short-term grazing impacts were not discernible. Annual vegetation may be influenced by grazing-induced defoliation via reduced seed production of preferred or more grazing-exposed species, or by favouring species with short life cycles, heterogeneous germination patterns or competitive advantages under low litter cover (Turner 1999 and references therein).

In systems dominated by perennial grasses, high grazing pressure can exacerbate drought mortality of grass tussocks and hinder post-drought establishment of seedlings (O'Connor 1991; 1994; O'Connor & Pickett 1992). Compositional changes and local extinction of grass species such as *Themeda triandra* following drought are greater under heavy grazing than under light or no grazing (O'Connor 1995; Fynn & O'Connor 2000). Perennial grasses invest less in reproduction from seed than annual grasses, and their dispersal, recruitment and establishment is therefore often seed-limited. As grass tufts die and grasses fail to re-establish, more soil becomes exposed and hence vulnerable to erosion. O'Connor and Roux (1995) found that the long-term response to grazing was most pronounced in longer-lived plants, whereas the growth of annual grasses directly responded to rainfall from year to year. More research is needed to gain a predictive understanding of the response of different types of vegetation communities to continuous high-intensity grazing.

In certain areas, long-term high grazing pressure has resulted in persistent and resilient vegetation assemblages dominated by grazing-resistant plant species. Examples are the perennial *Aristida junciformis* grasslands of Transkei in South Africa (McKenzie 1982) and annual grasslands with *Indigofera cliffordiana* dwarf shrubs in northern Kenya (Oba et al. 2000). Oba et al. (2000) found that high *Indigofera* mortality accompanied complete exclusion of ungulate herbivory for longer than five years, and this was followed by an increase of bare ground after eight years. The authors concluded that grazing was essential in maintaining the productivity and diversity of the vegetation and that lack of grazing, rather than overgrazing, leads to rangeland degradation. However, just as with conclusions about the effects of overgrazing, care must be taken with the definition and assessment of degradation, and when trying to extrapolate such findings to other regions.

Much of the heat of the debate has been generated by the different predictions of the two paradigms regarding degradation of arid and semi-arid rangelands. In some cases, the non-equilibrium paradigm has been embraced with such enthusiasm that concerns about degradation and the relevance of stocking rates were completely dismissed (for example, Dikeni et al. 1996 in South Africa). This has provoked criticism from ecologists concerned about the ecological consequences of uncritically adopting the non-equilibrium paradigm for management, for example in areas which they felt were not predominantly experiencing non-equilibrium dynamics (for example, Illius & O'Connor 1999; 2000; Fernandez-Gimenez & Allen-Diaz 1999; Cowling 2000; Desta & Coppock 2002). Areas where non-equilibrium concepts would be inappropriate include less drought-prone semi-arid rangelands at the more humid end of the spectrum, but also semi-arid, climatically variable areas where mobility of pastoralists has been severely restricted, or systems where the provision of seasonally scarce resources such as feed and water is reducing the temporal variation in animal growth even though rainfall and plant growth are low in drought years.

Supplementary feeding can increase drought survivorship and reduce the time before livestock can breed after drought stress. Buying in of livestock (and especially breeding stock) can speed up the recovery of the herd to its pre-drought size. For example, most livestock owners

in South African communal areas have cash incomes from migrant labour, local employment, remittances and/or pensions (Cousins 1998), making purchases of feed and stock possible. Data from a communal area in South Africa show that high livestock numbers are increasingly being maintained through the provision of feed and buying animals after droughts (Vetter & Bond 1999; Vetter 2003), and that livestock numbers thus remained high during and after droughts. Also in South Africa, data presented by Tapson (1993) show remarkably constant cattle numbers over a 25-year period, even during and after the severe drought of the early 1980s. An important research challenge is thus to understand the ecological consequences of restricting mobility in spatially heterogeneous areas, and of providing seasonally scarce resources such as water and feed in temporally variable environments.

MANAGEMENT OF EQUILIBRIUM AND NON-EQUILIBRIUM RANGELANDS

Planning and management of African communal rangelands has generally followed the equilibrium model and the assumption that these systems are overstocked and degraded. This has led to government interventions such as destocking schemes, conversion of communal areas into individually managed 'economic units' and settling of nomadic pastoralists into group ranches (Sandford 1983; Ellis & Swift 1988; Archer et al. 1989; Boonzaier et al. 1990; Rohde et al. 1999). The main focus of these interventions has been on preserving natural resources, with the additional intention of increasing livestock production and offtake, often for export or city markets. These schemes have met with widespread resistance, not least because they ignored the objectives of the pastoralists who derive a multitude of benefits from multi-species herds (Coughenour et al. 1985), many of which are non-consumptive (Shackleton et al. 2000). It is argued that these benefits are maximized at higher stocking rates than commercial farming objectives such as beef production (Sandford 1983; Abel & Blaikie 1989; Wilson & MacLeod 1991; Behnke & Abel 1996). Interventions often seemed to create or exacerbate, rather than solve, degradation problems and left many people economically worse off than before (Ellis & Swift 1988; Hoffman et al. 1999).

DEALING WITH TEMPORAL VARIABILITY AND DROUGHT

Commercial farmers in semi-arid environments, where droughts occur at regular intervals, generally maintain low enough stocking rates to ensure sufficient forage in years of low rainfall (at least this is the recommendation of agriculturalists; often commercial farmers keep higher than recommended stocking rates unless benefits such as drought subsidies are made conditional on maintaining conservative stocking rates). There is thus acknowledgement of climatic variability in equilibrium-based range management, but the proposed solution is to achieve stability by maintaining livestock at densities that are unlikely to exceed the reduced carrying capacity of dry years. However, it is argued that management based on constant and conservative stocking rates would be inappropriate and costly to pastoralists in such variable systems, as they would be unable to make use of all the available forage in wet years, and would still overstock in very dry years (Sandford 1982; 1983; Behnke & Scoones 1993). The opportunity cost of conservative stocking rates increases with increasing rainfall variability and more conservative stocking rates (Sandford 1982; 1983, Stafford Smith 1996).

Pastoralists employ a variety of strategies to cope with the variability of their environment (Sandford 1983; Ellis & Swift 1988; Scoones 1994). Instead of aiming to keep animal numbers constant, pastoralists allow herd size to change with rainfall (Sandford 1983; 1994; Toulmin 1994). Drought risks are minimised not by maintaining conservative stocking rates, but rather by allowing livestock numbers to increase in wet years. While livestock owners risk substantial losses during a severe drought, having a large herd at the beginning of the drought ensures that at least some part of the herd survives. The bigger the herd belonging to an individual in the communal system, the greater is the number likely to survive, and larger herds thus provide greater security during droughts.

The effectiveness of this strategy depends on how pre-drought livestock density affects livestock survival, condition and post-drought recovery, particularly of breeding females. If mortality is completely density-independent, the number of livestock before the drought does not affect the number that die (and survive), so keeping low livestock numbers will not reduce drought mortality. If, on the other hand, livestock mortality is density-dependent during drought, the effect of high pre-drought livestock numbers needs to be taken into consideration. Reducing livestock numbers before they reach densities where they exacerbate drought mortality, as suggested by Desta and Coppock (2002), would make sense under these conditions.

Management of livestock numbers in response to drought must take into account the variables of interest to pastoralists – that is, the benefits derived from the livestock herd in the years between droughts, and survival and recovery of the herd during and after the drought. To the livestock owner, the percentage mortality of the regional herd is not so much of interest as the number per household that survives the drought. If the number of livestock surviving a drought is the same regardless of initial density, and the benefits derived from livestock are proportional to livestock number, a strategy of maximising livestock numbers between droughts would be sensible. This results in a higher percentage of mortality, as well as a greater

number of livestock lost, but greater benefits derived between droughts and the same number of livestock after the drought. This scenario assumes that there are no short- or long-term effects on the vegetation if livestock numbers are high at the onset of drought. Short-term effects on vegetation productivity affect post-drought recovery of livestock, while long-term effects are of concern for the long-term sustainability of maintaining high stocking rates. Very often, there is a mismatch of the time-scale on which the benefits (short-term) and costs (long-term) of heavy grazing occur. In the short term, the benefits commonly exceed the costs, favouring the maintenance of high stocking rates even when there is a long-term risk of degradation (Ash et al. 2002).

It is argued that appropriate management in these highly variable systems should aim at supporting flexible responses to droughts, such as pre-empting drought mortality by marketing surplus animals, and offering opportunities to re-stock by buying in animals (Sandford 1983; Toulmin 1994; Behnke & Abel 1996). Opportunistic strategies are being recognised as better alternatives to constant, conservative stocking rates, even in commercial systems (Danckwerts et al. 1993). However, the economic efficiency and environmental sustainability of tight tracking strategies, particularly those that rely on buying stock after droughts, are still debated (for example, Sandford 1994; Illius et al. 1998; Campbell et al. 2000).

DEALING WITH SPATIAL HETEROGENEITY

Pastoral strategies to cope with climatic variability and drought also make use of spatial heterogeneity, as resource availability and rainfall are not evenly distributed across the landscape (Sandford 1983; Ellis & Swift 1988; Scoones 1995). Pastoralists in arid areas are usually fairly mobile and will move their animals to the best available grazing, covering different areas over the course of the year and between years. Such movements may be in the form of transhumance, which follows a more or less predictable pattern between wet and dry season resources, or more opportunistic movement tracking less predictable patterns of productivity, often caused by patchy rainfall patterns (Coughenour 1991; Bayer & Waters-Bayer 1994; Niamir-Fuller & Turner 1999; Fernandez-Gimenez & Swift 2003). Movement occurs at different temporal and spatial scales, some migrations involve certain subsets of the herds and households (Fernandez-Gimenez & Swift 2003). Neighbouring communities usually have arrangements for reciprocal grazing rights that pastoralists can make use of in bad years. These movement patterns are thought to enable farmers to maintain high stocking rates even in dry years without putting continuous pressure on the grazing resource throughout the year (Coughenour 1991; Ellis et al. 1993; Stafford Smith 1996).

Most rangelands also contain small, highly productive areas that make a disproportionately large contribution to the area's total forage production. Examples are riverine areas and drainage lines, which support green grass growth throughout most of the year, or croplands where animals can graze on crop residues in the dry season. Such 'key resource areas' are thought to play a vital role in carrying animals through the dry season bottleneck, and may be responsible for maintaining total animal numbers considerably higher than the predicted carrying capacities, which are based on a homogeneous landscape (Scoones 1993; 1995). In areas where semi-arid rangelands border on areas where crop production is possible, nomadic pastoralists and settled agriculturalists may have mutually beneficial arrangements where livestock use crop residues in the dry season, allowing the crop farmer to make use of manure. Where such relationships break down, the total number of livestock that can be kept may be considerably reduced as exploitation of the entire rangeland by pastoralists relies on mobility and access to crop residues in the dry season (Bayer & Waters-Bayer 1994). The same happens when the higher potential grazing areas are converted to cropland, forcing pastoralists into increasingly marginal land without access to key resources (Homewood & Rodgers 1987; Scoones 1990; Dodd 1994; Desta & Coppock 2002).

Ellis and Swift (1988), Scoones (1990:392), Ellis et al. (1993) and Bayer and Waters-Bayer (1994), among others, also stress that to persist through droughts, pastoralists need to be able to expand their operations into areas not normally used for grazing, and to gain access to outside resources. In household-level studies of livestock dynamics during drought, Homewood and Lewis (1987), Scoones (1990) and Oba (2001) found that mobility during droughts was a key factor contributing to herd survival. Even in the comparatively sedentary communal livestock farming systems in South Africa, there are reports of livestock owners gaining access to wetter areas far beyond their usual pastures in a devastating drought in 2003 (Alcock unpublished).

It seems to be widely accepted that the reduction of mobility in semi-arid and arid pastoral systems has increased the risk of degradation because of the way it concentrates grazing pressure on the resource and reduces the opportunities for resting parts of the vegetation (for example, Coughenour 1991; Perkins & Thomas 1993; Oba et al. 2000; Fernandez-Gimenez & Swift 2003). At the same time, remote areas become less frequently utilised and may lose productivity in the absence of periodic grazing (Niamir-Fuller 1999b). Constriction of mobility is associated with development interventions to settle nomadic pastoralists into ranches, encroachment of rangelands by other forms of land-use such as cultivation and conservation, increasing population densities in rangeland areas, and the proliferation of water points, often accompanied by

settlements. In sparsely populated arid areas, grazing impact is often concentrated in piospheres or 'sacrifice zones' around water points or settlements (Perkins & Thomas 1993; Sullivan 1999; Leggett et al. 2003), while the rest of the area is largely unaffected. In more densely populated rangelands, such as the former 'homelands' of South Africa where villages are few kilometres apart, high grazing pressure is much more continuous over the landscape.

There appears to be a need for developing management models which re-introduce mobility, to buffer pastoralists against temporal variability in forage availability, and to reduce localised degradation. When the traditional transhumant movements of cattle ranchers in the USA and South Africa became constricted by settled farmers early in the 20th century (Coughenour 1991), the solution to perceived degradation caused by the increasingly concentrated and continuous grazing pressure was the introduction of grazing systems such as rotational grazing and resting. These were intended to mimic the evolutionary grazing patterns by native ungulates, which consisted of intense, localized defoliation followed by periods of no grazing. However, in arid areas, movement in response to variable resource availability and drought is necessary on large scales and needs to be flexible. An alternative to rotational grazing and other forms of grazing management would be to restore mobility in rangelands. This would in many cases involve expanding the areas under communal tenure and re-establishing access to key resources, a strategy likely to clash with conservation agendas and other land users. Governments and many development agents have consistently perceived mobile peoples as at best inconvenient and unproductive, and at worst as a threat to 'civilized' settled land users. The legitimacy of mobility has been questioned and undermined in many countries, and reinstating mobility thus requires a fundamental change in mindset (Niamir-Fuller 1999b; Sullivan & Homewood in prep.)

SOME OBSTACLES AND THE WAY FORWARD

The resolution of the key issues in the debate, ecological dynamics, appropriate policy and management strategies and rangeland degradation, has been hindered by a number of factors. Firstly, different authors interpret 'non-equilibrium' and 'disequilibrium' in different ways, which can frustrate dialogue. To ecologists, the definition is very specific and refers to how strongly livestock population dynamics are coupled to plant production (for example, Illius & O'Connor 1999), and to the relative importance in grazing pressure and abiotic factors in determining vegetation composition and productivity (for example, Ellis & Swift 1988; Fernandez-Gimenez & Allen-Diaz 1999). Social scientists and more interdisciplinary researchers and practitioners tend to use a broader definition of non-equilibrium which encompasses the ecological, socio-economic, institutional and political complexities of living under environmental variability (for example, Sullivan & Rohde 2002; Sullivan & Homewood in prep.).

Much of the research and modelling that has been done to explore equilibrium vs non-equilibrium is not explicit about the type of pastoral system under question in terms of climate, principal economic and subsistence activities, mobility and access to outside resources. Recognising this problem, Stafford Smith (1996) suggested that a typology of rangeland types according to the above factors would lead to greater clarity when comparing results of different studies. He also pointed out the danger of broadly adopting a paradigm as dogma when even its proponents were explicit about the conditions under which it applies.

The debate about the ecological dynamics, sustainability and degradation in rangelands has been influenced substantially by the agendas of governments, development and influential conservation lobbies, and researchers may not always be aware of this. Most semi-arid and arid areas are not suitable for more intensive agriculture, and grazing by ungulates is the only way for humans to produce food in these systems. In such areas, however, there has been and continues to be, conflict with wildlife and biodiversity conservation, especially in eastern and southern Africa (for example, Homewood & Rodgers 1987; Little 1996; Bond 1999). In areas with high agricultural potential, the production of crops, often in irrigation schemes, has been held up as more productive and desirable, even though these forms of land use are often unsustainable and have much more devastating and permanent impacts on soil characteristics and biodiversity by completely transforming natural vegetation. Usually the viewpoint of the pastoralist or communal farmer is the least represented in these conflicts (Niamir-Fuller 1999a).

The lack of agreement on definitions of degradation and sustainability, and hence clear objectives and criteria for developing appropriate management has been a considerable impediment to an integrative effort at resolving the search for appropriate paradigms. Generally, there seems to have been a stand-off between supporters of the non-equilibrium paradigm (many of whom are social scientists) and its critics (many ecologists, range scientists and conservationists), and much of this appears to be based on ideology and personal values rather than data. Even where different people agree that degradation has occurred in some rangeland areas, their response to this depends on whether their priority is supporting pastoral livelihoods or preventing ecological degradation, and also what time scale they perceive as relevant for the analysis (for example, Abel 1993; Biot 1993; Reynolds & Stafford Smith 2002; Sullivan & Rohde 2002). Appropriate

management strategies in communal rangelands depend on ecological as well as socio-economic understanding of the systems and thus require researchers from these different disciplines to develop a common agenda and to integrate their efforts. To take the rangeland non-equilibrium debate forward in a meaningful way, researchers must progress from defending and verifying favoured theories to broad and objective testing of alternative hypotheses.

This has been hindered to some extent by the way in which the equilibrium and non-equilibrium viewpoints have been stereotyped. It has further polarised the debate and made the two positions appear irreconcilable. In the mainstream view, ‘equilibrium’ has been associated with settled commercial farming, private tenure and (and hence ‘good management’ based on hard science), and ‘non-equilibrium’ with communal, often mobile grazing (and hence ‘tragedy of the commons’). (For an in-depth discussion of the roots and consequences of this dichotomy see Sullivan & Homewood in prep.). The non-equilibrium paradigm has been portrayed as reckless and irresponsible in its views on degradation and its recommendations of opportunistic strategies (for example, Cowling 2000). On the other hand, proponents of the non-equilibrium paradigm have tended to caricature the equilibrium view as assuming constant climate and zero variability and promoting inflexible, euro-centric management strategies. A greater awareness of the context in which the two paradigms have evolved and a more open-minded engagement should hopefully help to close the ideological divide and open the way for objective research and constructive dialogue.

REFERENCES

Abel, N & Blaikie, P. 1989. Land degradation, stocking rates and conservation policies in the communal rangelands of Botswana and Zimbabwe. *Land Degradation and Rehabilitation,* 1:101–23.

Abel, N. 1993. Reducing cattle numbers of southern African communal range: Is it worth it? in *Range ecology at disequilibrium: New models of natural variability and pastoral adaptation in African savannas,* edited by R Behnke, I Scoones & C Kerven. London: Overseas Development Institute.

Alcock, C. 2003. *The hills are smelling: Situation report on stock losses, September/October 2003.* Unpublished report.

Archer, F, Hoffman, M & Danckwerts, J. 1989. How economic are the farming units of Leliefontein, Namaqualand? *Journal of the Grassland Society of Southern Africa,* 6:211–5.

Ash, A, McIvor, J, Corfield, J & Winter, W. 1995. How land condition alters plant-animal relationships in Australia’s tropical rangelands. *Agriculture, Ecosystems and Environment,* 56:77–92.

Ash, A, Stafford Smith, D & Abel, N. 2002. Land degradation and secondary production in semi-arid and arid grazing systems: What is the evidence? in *Global desertification: Do humans cause deserts?* edited by J Reynolds & D Stafford Smith. Berlin: Dahlem University Press.

Bayer, W & Waters-Bayer, A. 1994. Forage alternatives from range and field: Pastoral forage management and improvement in the African drylands, in *Living with uncertainty: New directions in pastoral development in Africa*, edited by I Scoones. London: Intermediate Technology.

Behnke, R & Abel, N. 1996. Revisited: The overstocking controversy in semi-arid Africa. *World Animal Review,* 87:3–27.

Behnke, R & Scoones, I. 1993. Rethinking range ecology: Implications for range management in Africa, in *Range ecology at disequilibrium: New models of natural variability and pastoral adaptation in African savannas,* edited by R Behnke, I Scoones & C Kerven. London: Overseas Development Institute.

Bell, R. 1982. The effect of soil nutrient availability on community structure in African ecosystems, in *Ecology of Tropical Savannas*, edited by B Huntley & B Walker. Berlin: Springer Verlag.

Biot, Y. 1993. How long can high stocking densities be sustained? in *Range Ecology at Disequilibrium: New models of natural variability and pastoral adaptation in African savannas,* edited by R Behnke, I Scoones & C Kerven. London: Overseas Development Institute.

Bond, W. 1999. Rangelands and biodiversity in southern Africa – impacts of contrasting management systems, in *People and rangelands: Building the future: Proceedings of the VI International Rangelands Congress, Townsville, Australia vol 1*, edited by D Eldridge & D Freudenberger:629–33.

Boonzaier, E, Hoffman, M, Archer, F & Smith, A. 1990. Communal land use and the ‘tragedy of the commons’: Some problems and development perspectives with reference to semi-arid regions of southern Africa. *Journal of the Grassland Society of Southern Africa,* 7:77–80.

Campbell, B, Doré, D, Luckert, M, Mukamuri, B & Gambiza, J. 2000. Economic comparisons of livestock production in communal grazing lands in Zimbabwe. *Ecological Economics,* 33:413–38.

Caughley, G. 1979. What is this thing called carrying capacity? in *North American elk: Ecology, behavior and management,* edited by M Boyce & L Hayden-Wing. Laramie, Wyoming: University of Wyoming Press.

Connell, J & Sousa, W. 1983. On the evidence needed to judge ecological stability or persistence. T*he American Naturalist,* 121:789–824.

Coppock, D. 1993. Vegetation and pastoral dynamics in the southern Ethiopian rangelands: Implications for theory and management, in *Range ecology at disequilibrium: New models of natural variability and pastoral adaptation in African savannas,* edited by R Behnke, I Scoones & C Kerven. London: Overseas Development Institute.

Coughenour, M. 1991. Spatial components of plant-herbivore interactions in pastoral, ranching, and native ungulate ecosystems. *Journal of Range Management,* 44:530–42.

Coughenour, M, Ellis, J, Swift, D, Coppock, D, Galvin, K, McCabe, J & Hart, T. 1985. Energy extraction and use in a nomadic pastoral ecosystem. *Science,* 20:619–25.

Cousins, B. 1996. *Range management and land reform policy in post-apartheid South Africa.* Cape Town: Programme for Land and Agrarian Studies, University of the Western Cape. (Land reform and agrarian change in southern Africa occasional paper; no. 2.)

Cousins, B. 1998. Invisible capital: The contribution of communal rangelands to rural livelihoods in South Africa, in *Communal rangelands in southern Africa: A synthesis of knowledge*, edited by T de Bruyn & P Scogings. Alice: Department of Livestock and Pasture Science, University of Fort Hare.

Cowling, R. 2000. Challenges to the 'new' rangeland science. *Trends in Ecology and Ecology and Evolution,* 15:303–4.

Danckwerts, J, O'Reagain, P & O'Connor, T. 1993. Range management in a changing environment: A southern African perspective. *Rangeland Journal,* 15:133–44.

DeAngelis, D & Waterhouse, J. 1987. Equilibrium and nonequilibrium concepts in ecological models. *Ecological Monographs,* 57:1–21.

Desta, S & Coppock, D. 2002. Cattle population dynamics in the southern Ethiopian rangelands, 1980–97. *Journal of Range Management,* 55:439–51.

Dikeni, L, Moorhead, R & Scoones, I. 1996. *Land use and environmental policy in the rangelands of South Africa: Case studies from the Free State and Northern Province.* Johannesburg: Land and Agricultural Policy Centre. (Working paper; no. 38.)

Dodd, J. 1994. Desertification and degradation in sub-saharan Africa: The role of livestock. *BioScience,* 44:28–34.

Dye, P & Spear, P. 1982. The effect of bush clearing and rainfall variability on grass yield and composition in south-west Zimbabwe. *Zimbabwe Journal of Agricultural Research,* 20:103–17.

Dyksterhuis, E. 1949. Condition and management based on quantitative ecology. *Journal of Range Management*, 2:104–15.

East, R. 1984. Rainfall, soil nutrient status and biomass of large African savanna mammals. *African Journal of Ecology,* 22:245–70.

Ellis, J. 1994. Climate variability and complex ecosystem dynamics: Implications for pastoral development, in *Living with uncertainty. New directions in pastoral development in Africa*, edited by I Scoones. London: Intermediate Technology.

Ellis, J, Coughenour, M & Swift, D. 1993. Climate variability, ecosystem stability, and the implications for range and livestock development, in *Range ecology at disequilibrium: New models of natural variability and pastoral adaptation in African savannas,* edited by R Behnke, I Scoones & C Kerven. London: Overseas Development Institute.

Ellis, J & Swift, D. 1988. Stability of African pastoral ecosystems: Alternate paradigms and implications for development. *Journal of Range Management,* 41:450–9.

Fernandez-Gimenez, M & Allen-Diaz, B. 1999. Testing a non-equilibrium model of rangeland vegetation dynamics in Mongolia. *Journal of Applied Ecology,* 36:871–85.

Fernandez-Gimenez, M & Swift, D. 2003. Strategies for sustainable grazing management in the developing world, in *Proceedings of the VIIth International Rangelands Congress, Durban, South Africa.*

Foran, B, Tainton, N & Booysen, P. 1978. The development of a method for assessing veld condition in three grassland types in Natal. *Proceedings of the Grassland Society of Southern Africa,* 13:27–33.

Friedel, M. 1991. Range condition assessment and the concept of thresholds: A viewpoint. *Journal of Range Management,* 44:422–6.

Fritz, H & Duncan, P. 1994. On the carrying capacity for large ungulates of African savanna ecosystems. *Proceedings of the Royal Society of London*, 256:77–82.

Fynn, R & O'Connor, T. 2000. Effect of stocking rate and rainfall on rangeland dynamics and cattle performance in a semi-arid savanna, South Africa. *Journal of Applied Ecology,* 37:491–507.

Hardin, G. 1968. The tragedy of the commons. *Science,* 162:1243–8.

Heady, H & Child, D. 1994. *Rangeland ecology and management.* Boulder, Colorado: Westview Press.

Hoffman, M, Todd, S, Ntshona, Z & Turner, S. 1999. *Land degradation in South Africa.* Final report to the Department of Environmental Affairs and Tourism, South Africa.

Homewood, K & Lewis, J. 1987. Impact of drought on pastoral livestock in Baringo, Kenya 1983–85. *Journal of Applied Ecology,* 24:615–31.

Homewood, K & Rodgers, W. 1987. Pastoralism, conservation and the overgrazing controversy, in *Conservation in Africa: People, policies and practice*, edited by D Anderson & R Grove. Cambridge: Cambridge University Press.

Illius, A, Derry, J & Gordon, I. 1998. Evaluation of strategies for tracking climatic variation in semi-arid grazing systems. *Agricultural Systems*, 57:381–98.

Illius, A & O'Connor, T. 1999. On the relevance of non-equilibrium concepts to semi-arid grazing systems. *Ecological Applications*, 9:798–813.

Illius, A & O'Connor, T. 2000. Resource heterogeneity and ungulate population dynamics. *Oikos*, 89:283–94.

Jones, R & Sandland, R. 1974. The relation between animal gain and stocking rate: Derivation of the relation from the results of grazing trials. *Journal of Agricultural Science,* 83:335–41.

Kelly, R & Walker, B. 1976. The effects of different forms of land use on the ecology of a semi-arid region in south-eastern Rhodesia. *Journal of Ecology,* 64:553–76.

Lamprey, H. 1983. Pastoralism yesterday and today: the overgrazing controversy, in *Tropical savannas: Ecosystems of the world, vol. 13*, edited by F Bourlière. Amsterdam: Elsevier.

Leach, M & Mearns, R. (eds). 1996. *The lie of the land: Challenging received wisdom on the African environment*. Cambridge: Cambridge University Press.

Leggett, K, Fennessy, J & Schneider, S. 2003. Seasonal vegetation changes in the Hoanib River catchment, north-western Namibia: A study of a non-equilibrium system. *Journal of Arid Environments,* 53:99–113.

Little, P. 1996. Pastoralism, biodiversity, and the shaping of savanna landscapes in East Africa. *Africa,* 66:37–51.

McKenzie, B. 1982. Stability and resilience of the grasslands of Transkei. *Proceedings of the Grassland Society of Southern Africa,* 17:21–4.

Niamir-Fuller, M. 1999a. International aid for rangeland development: Trends and challenges, in *People and rangelands: Building the future: Proceedings of the VI International Rangelands Congress, Townsville, Australia vol 1*, edited by D Eldridge & D Freudenberger:147–53.

Niamir-Fuller, M. 1999b. Introduction, in *Managing mobility in African rangelands: The legitimization of transhumance*, edited by M Niamir-Fuller. London: Intermediate Technology.

Niamir-Fuller, M & Turner, M. 1999. A review of recent literature on pastoralism and transhumance in Africa, in *Managing mobility in African rangelands: The legitimization of transhumance*, edited by M Niamir-Fuller. London: Intermediate Technology.

O'Connor, T. 1985. *A synthesis of field experiments concerning the grass layer in the savanna regions of southern Africa.* Pretoria: Council for Scientific and Industrial Research. (South African National Scientific Programmes report; no. 114.)

O'Connor, T. 1991. Local extinction in perennial grasslands: A life-history approach. *The American Naturalist,* 137:753–73.

O'Connor, T. 1994. Composition and population responses of an African savanna grassland to rainfall and grazing. *Journal of Applied Ecology,* 31:155–71.

O'Connor, T. 1995. Transformation of a savanna grassland by drought and grazing. *African Journal of Range and Forage Science,* 12:53–60.

O'Connor, T. & Pickett, G. 1992. The influence of grazing on seed production and seed banks of some African savanna grasslands. *Journal of Applied Ecology,* 29:247–60.

O'Connor, T & Roux, P. 1995. Vegetation changes (1949–71) in a semi-arid, grassy dwarf shrubland in the Karoo, South Africa: Influence of rainfall variability and grazing by sheep. *Journal of Applied Ecology*, 32:612–26.

Oba, G. 2001. The effects of multiple droughts on cattle in Obbu, Northern Kenya. *Journal of Arid Environments,* 49:375–86.

Oba, G, Stenseth, N & Lusigi, W. 2000. New perspectives on sustainable grazing management in arid zones of sub-saharan Africa. *BioScience,* 50:35–51.

Owen-Smith, R. 2002. *Adaptive herbivore ecology: From resources to populations in variable environments.* Cambridge: Cambridge University Press.

Perkins J & Thomas, D.1993. Spreading deserts or spatially confined environmental impacts? Land degradation and cattle ranching in the Kalahari desert of Botswana. *Land Degradation and Rehabilitation,* 4:179–94.

Phelps, D & Bosch, O. 2002. A qualitative state and transition model for the Mitchell grasslands of central western Queensland. *Rangeland Journal*, 24:242–67.

Reynolds, J & Stafford Smith, D. 2002. Do humans cause deserts? *Global desertification: Do humans cause deserts?* Edited by J Reynolds & M Stafford Smith. Berlin: Dahlem University Press.

Rohde, R, Hoffman, T & Cousins, B. 1999. *Experimenting with the commons: A comparative history of the effects of land policy on pastoralism in two former 'reserves' in Namibia and South Africa.* Cape Town: Programme for Land and Agrarian Studies, University of the Western Cape. (Land reform and agrarian change in southern Africa occasional paper; no. 12.)

Sandford, S. 1982. Pastoral strategies and desertification: Opportunism and conservatism in dry lands, in *Desertification and development: Dryland ecology in social perspective*, edited by B Spooner & H Mann. London: Academic Press.

Sandford, S. 1983. *Management of Pastoral Development in the Third World.* New York: John Wiley and Sons.

Sandford, S. 1994. Improving the efficiency of opportunism: New directions for pastoral development, in *Living with uncertainty: New directions in pastoral development in Africa*, edited by I Scoones. London: Intermediate Technology.

Scoones, I. 1990. *Livestock populations and the household economy: A case study from southern Zimbabwe.* London: Unpublished PhD thesis, University of London.

Scoones, I. 1993. Why are there so many animals? Cattle population dynamics in the communal areas of Zimbabwe, in *Range ecology at disequilibrium: New models of natural variability and pastoral adaptation in African savannas*, edited by R Behnke, I Scoones & C Kerven. London: Overseas Development Institute.

Scoones, I. 1994 (ed.). *Living with Uncertainty. New directions in pastoral development in Africa*. London: Intermediate Technology.

Scoones, I. 1995. Exploiting heterogeneity: Habitat use by cattle in dryland Zimbabwe. *Journal of Arid Environments*, 29:221–37.

Scoones, I. 1999. New ecology and the social sciences: What prospects for a fruitful engagement? *Annual Review of Anthropology*, 28:479–507.

Shackleton, S, Shackleton, C & Cousins, B. 2000. *Re-valuing the communal lands of southern Africa: New understandings of rural livelihoods.* London: Overseas Development Institute. (ODI Natural Resource Perspectives; no. 62).

Stafford Smith, M. 1996. Management of rangelands: Paradigms at their limits, in *The ecology and management of grazing systems*, edited by J Hodgson & A Illius. Wallingford, Oxfordshire: CAB International.

Sullivan, S. 1996. Towards a non-equilibrium ecology: Perspectives from an arid land. *Journal of Biogeography,* 23:1–5.

Sullivan, S. 1999. The impacts of people and livestock on topographically diverse open wood- and shrublands in arid north-west Namibia. *Global Ecology and Biogeography Letters*, 8:257–77.

Sullivan, S & Homewood, K. (in prep.) On non-equilibrium and nomadism: Knowledge, diversity and modernity in drylands, in *Reclaiming knowledge for diversity*, by M. Pimbert. London: Earthscan.

Sullivan, S & Rohde, R. 2002. On non-equilibrium in arid and semi-arid grazing systems. *Journal of Biogeography,* 29:1595–618.

Tainton, N. 1981. The ecology of the main grazing lands of South Africa, in *Veld and pasture management in South Africa*, edited by N Tainton. Pietermaritzburg: Shuter and Shooter.

Tainton, N. (ed.). 1999. *Veld management in South Africa*. Pietermaritzburg: University of Natal Press.

Tapson, D. 1993. Biological sustainability in pastoral systems: The Kwazulu case, in *Range Ecology at disequilibrium: New models of natural variability and pastoral adaptation in African savannas*, edited by R Behnke, I Scoones & C Kerven. London: Overseas Development Institute.

Toulmin, C. 1994. Tracking through drought: Options for destocking and restocking, in *Living with uncertainty. New directions in pastoral development in Africa*, edited by I Scoones. London: Intermediate Technology.

Trollope, W. 1990. Development of a technique for assessing veld condition in the Kruger National Park. *Journal of the Grassland Society of Southern Africa*, 7:46–51.

Turner, M. 1999. Spatial and temporal scaling of grazing impact on the species composition and productivity of Sahelian annual grasslands. *Journal of Arid Environments*, 41:277–97.

Vetter, S. 2003. *What are the costs of land degradation to communal livestock farmers in South Africa? The case of the Herschel district, Eastern Cape*. Cape Town: Unpublished PhD thesis, University of Cape Town.

Vetter, S & Bond, W. 1999. What are the costs of environmental degradation to communal livestock farmers? in Rangelands and biodiversity in southern Africa – impacts of contrasting management systems, in *People and rangelands: Building the future: Proceedings of the VI International Rangelands Congress, Townsville, Australia vol 1*, edited by D Eldridge & D Freudenberger:537–8.

Ward, D, Ngairorue, B, Kathena, J, Samuels, R & Ofran, Y. 1998. Land degradation is not a necessary outcome of communal pastoralism in arid Namibia. *Journal of Arid Environments*, 40:357–71.

Ward, D, Ngairorue, B, Karamata, J & Kapofi, I. 2000. The effects of communal and commercial pastoralism on vegetation and soils in an arid and a semi-arid region of Namibia, in *Vegetation science in retrospect and perspective*, edited by P White, L Mucina, J Leps & E van der Maarel. Uppsala, Sweden: Opulus Press.

Westoby, M, Walker, B & Noy-Meir, I. 1989. Opportunistic management for rangelands not at equilibrium. *Journal of Range Management*, 42:266–74.

Wiens, J. 1977. On competition and variable environments. *American Scientist*, 65:590–7.

Wiens, J. 1984. On understanding a non-equilibrium world: Myth and reality in community patterns and processes, in *Ecological communities: Conceptual issues and the evidence*, edited by D Strong, D Simberloff, L Abele, & A Thistle. Princeton, New Jersey: Princeton University Press.

Wiens, J. 1989. *The ecology of bird communities: Processes and variations*, vol 2. Cambridge: Cambridge University Press.

Wilson, A & MacLeod, N. 1991. Overgrazing: Present or absent? *Journal of Range Management*, 44:475–82.

THE DEFINITION OF NON-EQUILIBRIUM AND THE ROLE OF KEY RESOURCE – AN ECOLOGICAL PERSPECTIVE

ANDREW ILLIUS AND TIM O'CONNOR

The important role played by temporal and spatial heterogeneity in the dynamics of semi-arid grazing systems is well recognised. Highly variable rainfall causes episodic drought-induced mortality in herbivore populations. Livestock populations exploit spatial variation in resource abundance and are dependent on 'key resource' areas during the dry season. In contrast to the earlier view that plants and animals exist in some sort of equilibrium, it is now argued that, due to environmental variability, their populations are governed by fundamentally different, 'non-equilbrial' processes in which plant and animal dynamics are largely independent of each other. This paper seeks to clarify these concepts, in ecological terms.

We can define 'key resources' in relation to the key factor. The key factor determining animal population size is survival over the season of plant dormancy, so key resources are those eaten during this time. In other words, we can posit that key resources limit population size *via* the key factor, and that a reduction in these resources would cause the population to decline. Modelling results show that this is indeed the case: long-term mean animal abundance was largely determined by the quantity of resources available during the dry season, when the key factor of mortality operates, and scarcely at all by resources available in the wet season.

Environmental variability disturbs the equilibrium that could be reached between consumers and resources under stable conditions. This condition of disequilibrium, arising from climatic variation, is different from non-equilibrium, which could be defined as the absence of coupling between the animal population's dynamics and the subset of resources not associated with key factors. Wet season rangeland can therefore be classed as a non-equilibrium resource, because the animal population's dynamics are not coupled to it. Superabundance of non-key resources is likely to be observed during the growing season, because the animal population is typically limited by scarcer, high quality resources during the dormant season. Diet selection from heterogeneous resources will naturally cause the animal population's dynamics to depend differentially on different resources. But this is not primarily the consequence of climatic variability. In addition, we cannot characterise entire grazing systems in highly variable climates as 'non-equilibrial'.

The extent to which the non-equilibrium part of consumer-resource systems is prone to impact depends on the relative abundance of the two resource types, because animal numbers are regulated by key resources and are not coupled to non-equilibrium resources. Thus, high ratios of key non-equilibrium resources could support animal populations large enough to result in relatively high defoliation intensities of the non-equilibrium resources. An extreme case of this effect would occur if animals were maintained on supplementary food over the dry season. Then, their numbers would tend to become completely uncoupled from range resources, and defoliation intensity of the wet season range would be a function of the numbers maintained.

UNDERSTANDING ARID AND SEMI-ARID GRAZING SYSTEMS: WHY BOTH EQUILIBRIUM AND NON-EQUILIBRIUM CONCEPTS ARE INADEQUATE

David Richardson, Brian Hahn and Timm Hoffman

INTRODUCTION

Discussion of the relevance of non-equilibrium theory to grazing systems in areas of highly variable rainfall has been based on conceptual models. In contrast to the earlier view that herbivores and plants exist in some sort of equilibrium, there is an argument that in these environments forage production is largely determined by rainfall and is unaffected by animal population density. This is because severe mortality during droughts keeps livestock densities below equilibrium (Ellis & Swift 1988; Behnke & Scoones 1993). This leads to the assumption that grazing has limited effect on long-term forage production. Recently, Illius and O'Connor (1999) questioned the relevance of non-equilibrium concepts to arid grazing systems. Subsequently Sullivan and Rohde (2002) re-examined the evidence considered by Illius and O'Connor and concluded that 'non-equilibrium theory provides a powerful explanatory model of pastoral eco- and social system dynamics' (but see also Briske et al. 2003, who have described an alternative approach to the definition of non-equilibrium, and Bayer & Waters-Bayer, this volume, who have pointed out that the dynamics of arid rangeland systems are probably more complex than either the equilibrium or non-equilibrium theories suggest). In this paper two long-term models (standard model and simplified model) are used to simulate the semi-arid shrubland pastoral system at Paulshoek in Namaqualand (South Africa) and to examine the applicability of equilibrium and non-equilibrium theory to semi-arid rangelands.

Our models assume four plant guilds which are dominant in Namaqualand and which have been generally described (Todd & Hoffman 1999) as woody perennial shrubs palatable to livestock (WP), succulent shrubs, primarily in the family Mesembryanthemaceae (ME), an unpalatable shrub guild which is dominated by *Galenia africana* (kraalbos) (GA) and annuals and geophytes (AG). Fundamental to development of the standard model is the recognition of two different characteristics of the vegetation: the population and biomass of each guild. Furthermore only a fraction of the biomass of each guild is edible.

Rangelands are complex systems with many components (Figure 1). Before sustainable management strategies can be developed, the inter-relations between the components and how they are influenced by rainfall, both in the short- and long-term, need to be fully understood. Mathematical models enable integration of knowledge about all the individual processes within the system. A hierarchy of simulation models has been developed for Paulshoek, a communal grazing area of 20 000ha in the Leliefontein Communal Reserve. This area has carried on average a total of 3 550 sheep and goats during the past 30 years, which is 1.775 times the stocking rate recommended by the Department of Agriculture. The mean annual rainfall is 190mm but is highly variable and most of the rain is during the winter months. This area was chosen as a long-term data set of animal numbers and detailed vegetation surveys were available. Stock numbers varied widely between years and the performance of individual animals varies widely within any year. As very few observations are available for many of the processes within the system (amounts of different plant species eaten, animal growth, reproduction and milk yields) a two-stage approach to modelling the system was adopted. First, a mechanistic model of individual plants and animals based on published information and established theory was developed with an integration interval of 0.1 day. The short time step enabled the simulation of the effects of hand milking suckling female goats (does) on milk production and kid growth. This model was then used to generate appropriate information for use in a long-term ecosystem model (Figure 2). Woody perennials and the annuals provide the most palatable and digestible forage. As 63% of the livestock in Paulshoek are goats, model output presented in this paper will be for goats.

Figure 1: Components of rangeland systems and models

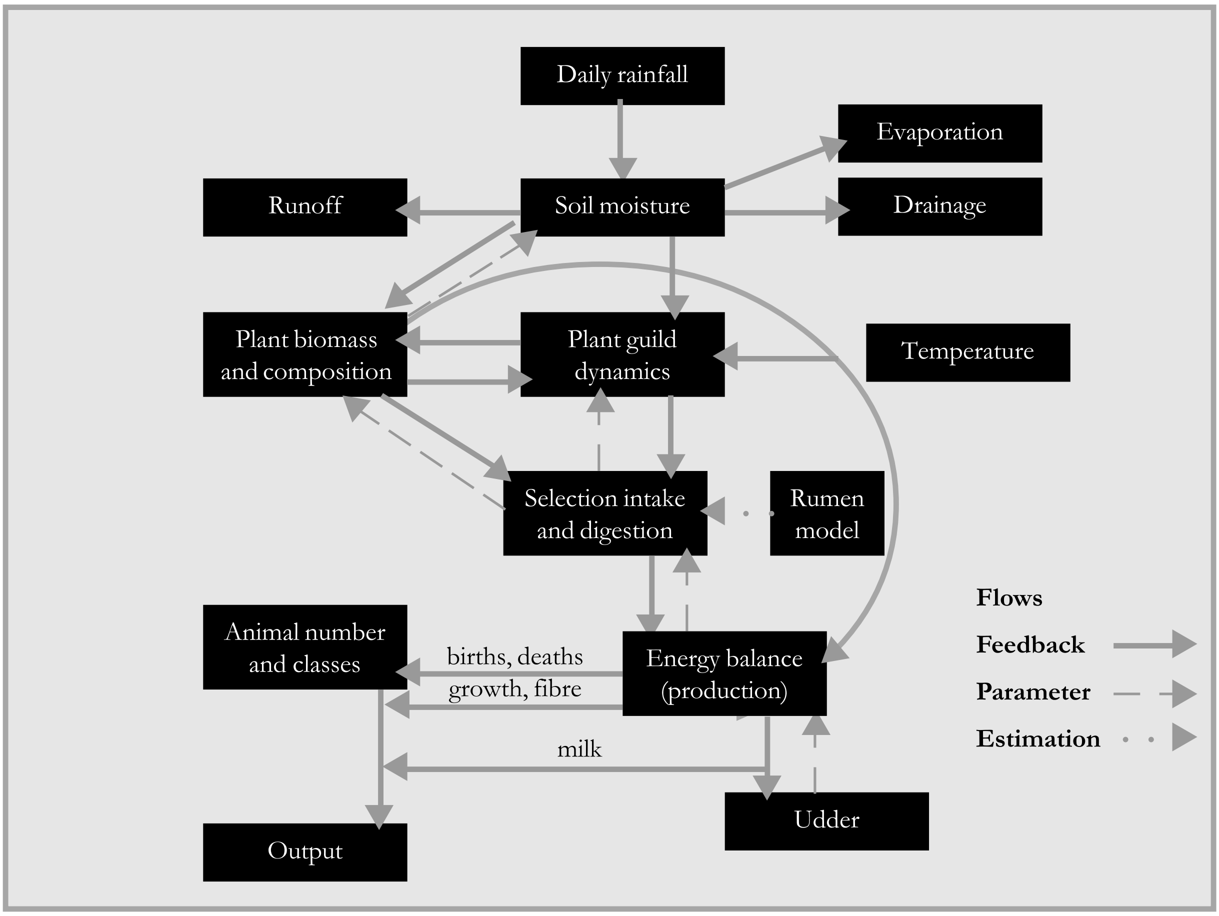
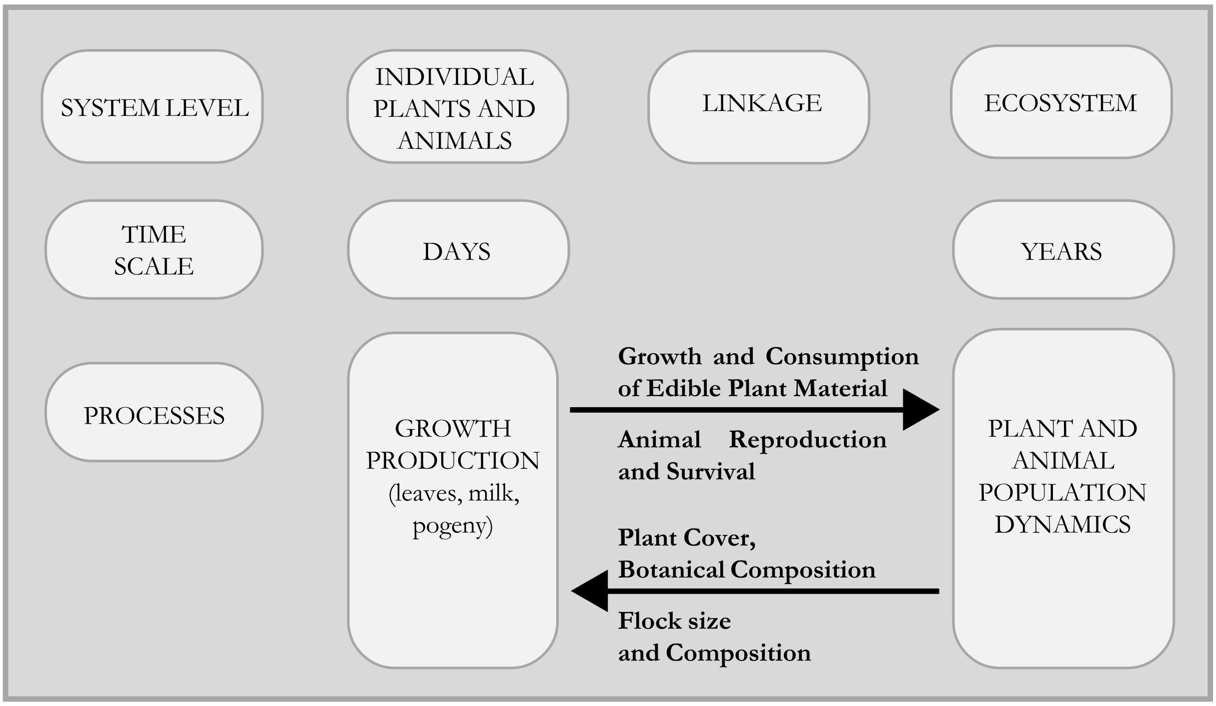


Figure 2: Relations between mechanistic and ecosystem models



COMPARISON BETWEEN SIMULATED AND OBSERVED VALUES

The mechanistic model was run using the recorded daily rainfall data for 2000/01 and the stocking rate was set to the recorded rate of 0.145 goats/ha. The model simulated does of different live weights at conception (31; 45 or 70kg) rearing either singles or twins. The does were not hand-milked. The simulated patterns of growth track the observed data well (Figure 3), supporting the concepts used in the model that doe weights influence foetal growth, milk production and growth of their kids. Differences in doe weight at conception are the result of number of kids reared in the previous year and genetic differences in mature size.

APPLICATION OF MECHANISTIC MODEL

A series of experiments was performed using the mechanistic model in the same way as one would carry out field trials, but with a far wider combination of treatments and conditions than would be possible with conventional experimentation. Growing animals, barren adults and does rearing singles or twins were simulated in factorial combination with stocking rates ranging from 0.05 to 0.5 small stock units (SSU)/ha. As the amount and temporal distribution of rainfall vary between years, many different years were simulated using daily rainfall records as input. Paulshoek data were used for the years that they are available; otherwise rainfall records from the town of Springbok were used. Variation in range condition was accommodated by runs with different populations of WP, ME and GA.

The effects of stocking rate, rainfall and vegetation condition can be combined to form one variable (forage availability) for each guild. Edible WP is used as an example as it forms an important part of the animal's diet and the amount eaten has a major effect on animal performance. The availability of edible WP per goat (WP/Goat) is calculated from yield of edible WP per ha on ungrazed veld (WP/ha) in that year and stocking rate as goats per ha (goats/ha)

Equation 1:
WP/Goat = (WP/ha) / (goats/ha)

Model output for twin kids whose dams were not milked shows that kid liveweight at six months of age is closely related to WP availability but decreases the later the onset of the rainy season (Figure 4). Some of the variation in kid weight is due to differences in initial doe weight and differences between years in rainfall pattern not accounted for by differences in date of the start of the season.

The amount of WP eaten by a breeding doe (WPeat, kg/year) is influenced by the availability of WP, availability of annuals (AG/goat), availability of dead WP (DWP), season (Se) and number of kids suckled (Equation 2). A reason for the effect of season of rainfall on the amount eaten is the length of time during which the goat has access to adequate amounts of edible WP.

Equation 2:
WPeat = 749.6 – 419.1exp (-0.00 219WP/goat)
– 80.55Log (AG/goat) + 2.837Log(DWP/goat)
– 8.042Se + 18.45Kids

Figure 3: Observed and predicted kid weights: 2000/01

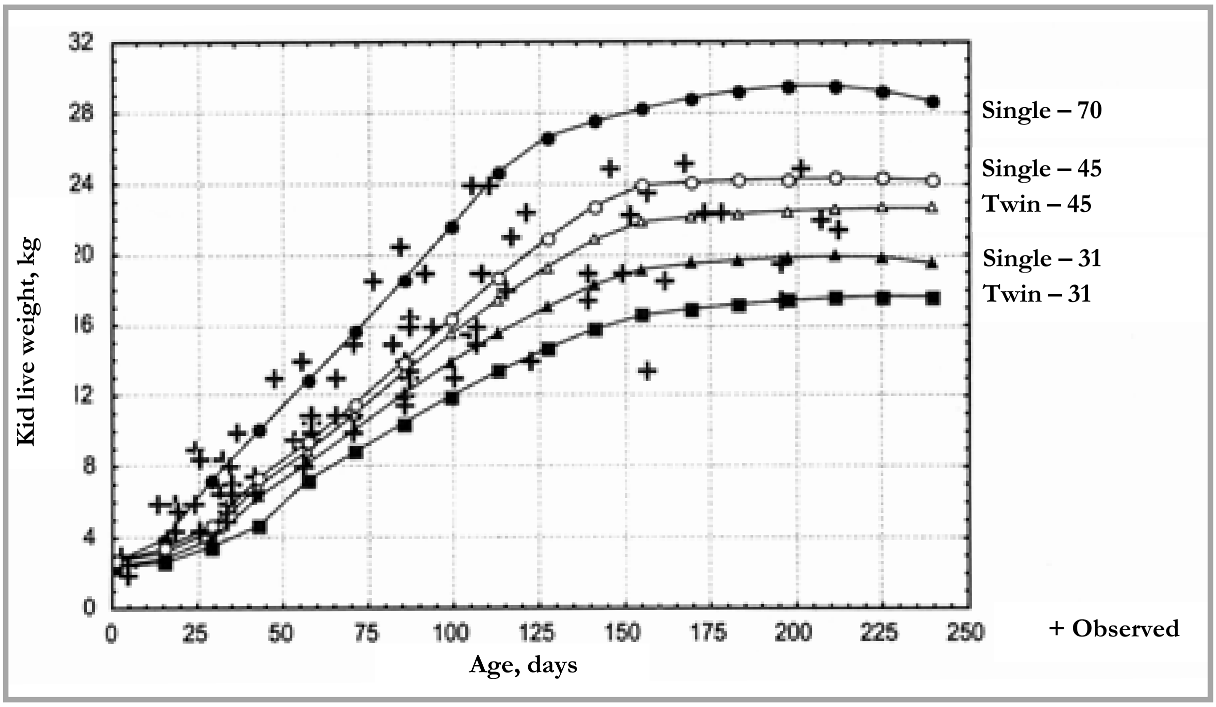
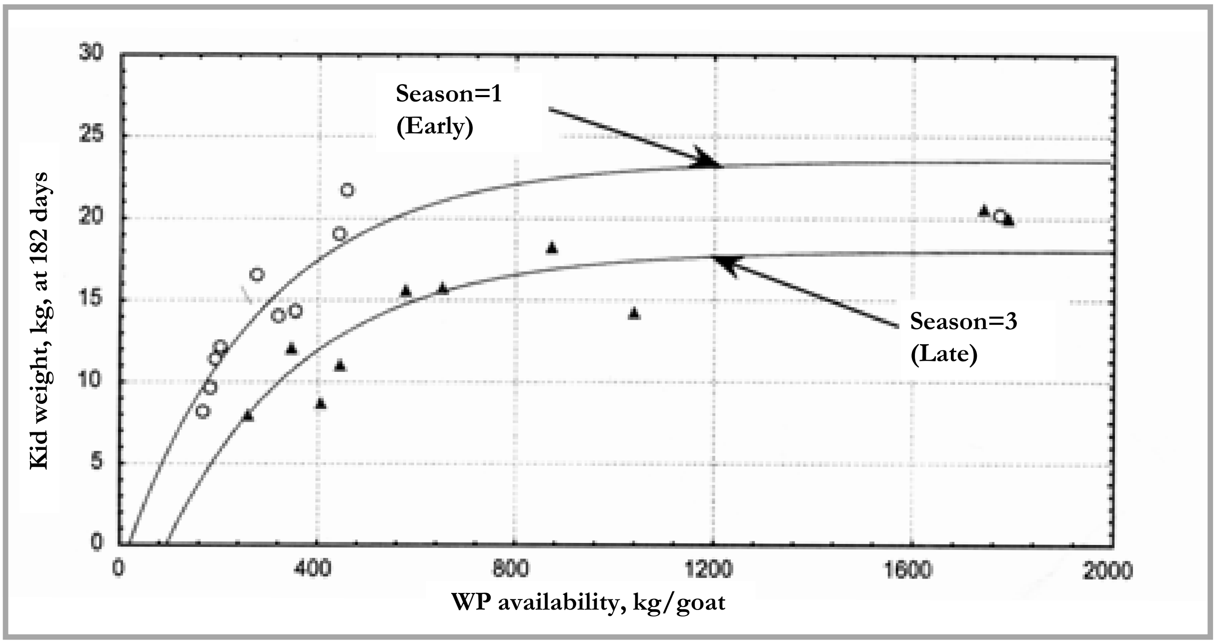


Figure 4: Effect of WP availability and rainfall season on kid weight



The weight of kids (Wt_{kid}) at weaning predicted by the mechanistic model is related to WP/Goat and to rainfall season (Se), number of kids reared (Kids), doe liveweight at mating (Doein) and whether the dam is hand milked or not (Milk) (Equation 3).

Equation 3:

$$\mathbf{Wt_{kid} = 26.79 - 28.12exp(-0.0\ 047\ WP/Goat) - 2.9Se + 0.193Doein - 4.53Kids - 2.08Milk}$$

Livestock reproductive and mortality rates in individual years have a major impact on the long-term performance of the system. A study of mechanistic model output indicates that reproductive rate is related to live weight at mating time and to the availability of WP. Doe survival rate (SR_{doe}) is directly related to live weight (Wt_{doe}) at the end of the dry season (Equation 4).

Equation 4:

$$\mathbf{SR_{doe} = 0.958\ 135 - 68\ 181.2\ exp\ (-0.35\ 454\ Wt_{doe})}$$

Analysis of model output showed that survival of kids (SR_{kid}) to weaning is not closely related to weight at any age, but rather to the amount of milk consumed which determines the rate of growth of very young kids. It was predicted that most kid deaths would occur in the first eight weeks of life. Rainfall season was also shown to be a factor affecting kid survival (Equation 5).

Equation 5:

$$\mathbf{SR_{kid} = 1.0\ 835 - 5.0\ 879exp(-0.0\ 9362Milk/kid) - 0.0\ 529Se}$$

Sets of equations and rules developed in this way are used to model behaviour of the system over decades. The models simulate the present Paulshoek production system, in which all male animals are sold or slaughtered before they are one year old and any does that survive until eight years old are slaughtered. Two such long-term models have been developed: the 'simplified' model and the 'standard' model.

SIMPLIFIED LONG-TERM MODEL

For this model the edible material of the three palatable guilds (WP, ME and AG) is aggregated into one variable: Forage. The toxic unpalatable shrub *Galenia africana* is excluded as it forms only a small fraction of forage eaten. The goats are divided into two age classes, adults and yearlings. As the simulated year starts at conception and kids are weaned at the end a simulated adult comprises a doe and her kids if any. The estimated amount of forage eaten by sucking kids is included in the estimate of the quantity consumed by the doe.

Forage growth depends on present forage density (F) and maximum potential forage density (F_{max}) (Equation 6). In the equilibrium version of the model F_{max} is set at a constant value and when variable rainfall is simulated F_{max} is a function of rainfall (R) and potential forage density when rainfall is optimal (F_{pot}) (Equation 7).

Equation 6:

$$\mathbf{dF/dt = a_F\ F\ (1 - F/F_{max}) - F_{eat}tot\ /A}$$

Equation 7:

$$\mathbf{F_{max} = ac/[\{a^b + (c^b - a^b)e^{(-0.012431R)}\}^{(1/b)}]F_{pot}}$$

APPLICATION OF THE SIMPLIFIED MODEL:

AN EQUILIBRIUM GRAZING SYSTEM WHERE SOIL MOISTURE AND TEMPERATURE DO NOT LIMIT PLANT GROWTH

As the objective of this paper is to examine the applicability of equilibrium concepts to semi-arid regions

and to Namaqualand in particular, the model was first run with the potential yield of forage (F_{max}) set at a constant value of 200kg/ha so that it could be used to simulate an equilibrium grazing system. In such a system, soil moisture, ambient temperature and soil fertility do not limit forage production. Consequently, the productivity of both vegetation and livestock depend on the numbers, sizes and classes of grazing animals. These characteristics do not apply to semi-arid rangeland, but the simplified model is used in this way to show that it is an equilibrium model and then to develop the argument that semi-arid shrubland does not exhibit equilibrium dynamics.

The area simulated was 20 000ha and the initial numbers of animals were set at 1 600 head (1 200 does and 400 yearlings). Both the density of edible forage and number of goats oscillated and the amplitude of the oscillations decreased over time until the system reached equilibrium after 20 years. The variation in stock numbers followed a similar pattern to the changes in forage but lagged by up to four years (Figure 5). This behaviour is typical of an equilibrium plant herbivore system (Crawley 1983). At equilibrium, edible forage density was 116.3kg/ha and the number of does plus yearlings was 4 139.

SIMULATION OF THE EFFECTS OF VARIABLE RAINFALL

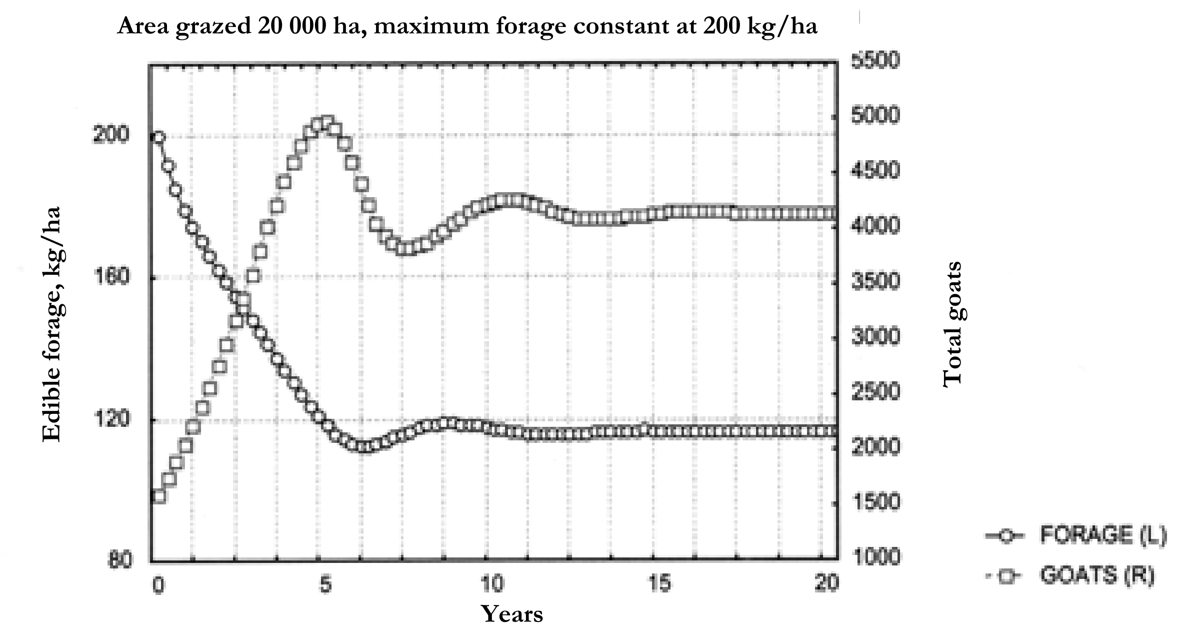
The effects of variable rainfall were simulated by making F_{max} vary with rainfall (Equation 7). So that the simplified ‘equilibrium’ model may be compared with the standard model, both models have been run using the recorded rainfall for the years 1971–2000. Both forage density and stock numbers responded to the variation in rainfall, but the pattern of changes in stock numbers lagged a year behind changes in forage production (Figure 6). Although the temporal behaviour of the system appears to be similar to that of the recorded data and the output of the standard model when the same rainfall data set is used (Figure 7) the similarity is superficial. The variation in goat numbers is tightly coupled with the variation in forage density from year four onwards. The independence of stock numbers from vegetation during the first four years is a consequence of the relatively low initial numbers (1 600 non-kids).

STANDARD LONG-TERM MODEL

Single replicates of the long-term model described by Hahn et al. (2003) (the standard model) are used to examine the response of the system over time to different management strategies and to differences in initial vegetation condition when subjected to the same rainfall pattern. The standard rainfall pattern comprises the 30 years of recorded rainfall data for Paulshoek, followed by 70 years of randomly generated rainfall having the same mean and standard deviation as the recorded data. The effects of different rainfall patterns on the behaviour of the system when subjected to the same management decisions are also studied. The initial vegetation conditions and livestock numbers are:

WP cover	20%
ME cover	12%
GA cover	6%
Mature Does	1 200
Yearling Does	400

Figure 5: Variation in forage density and goat numbers



Area grazed 20 000 ha, maximum forage constant at 200 kg/ha

Figure 6: Variation in forage density and goat numbers

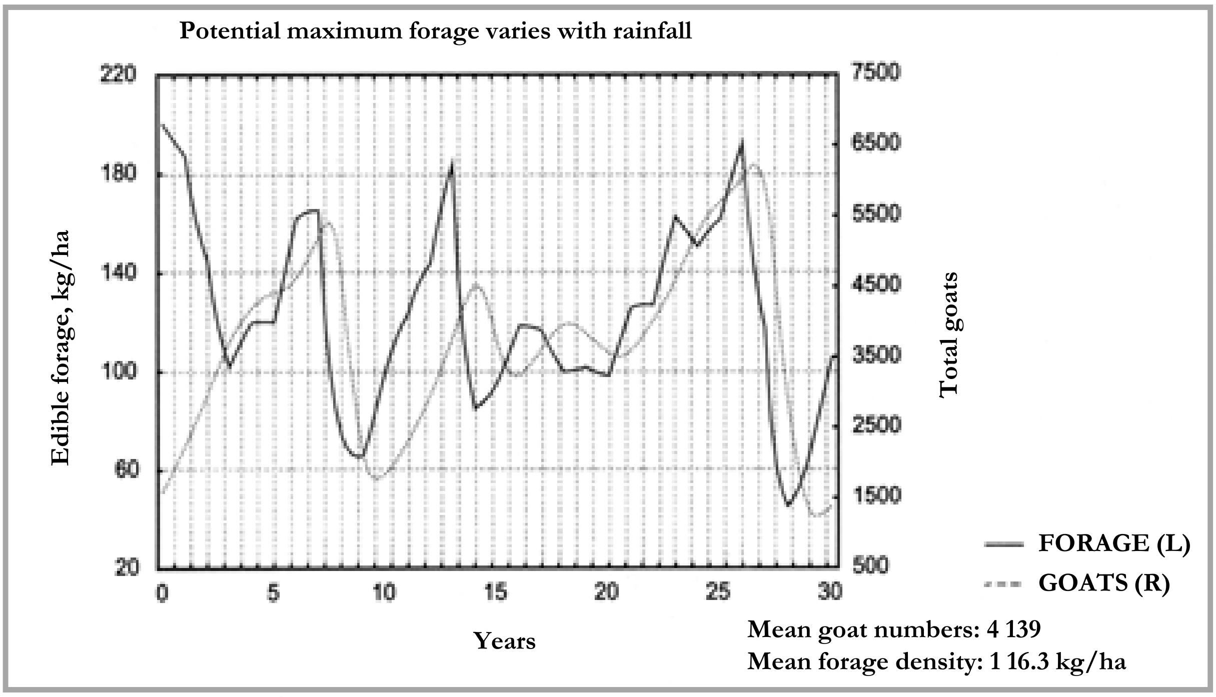
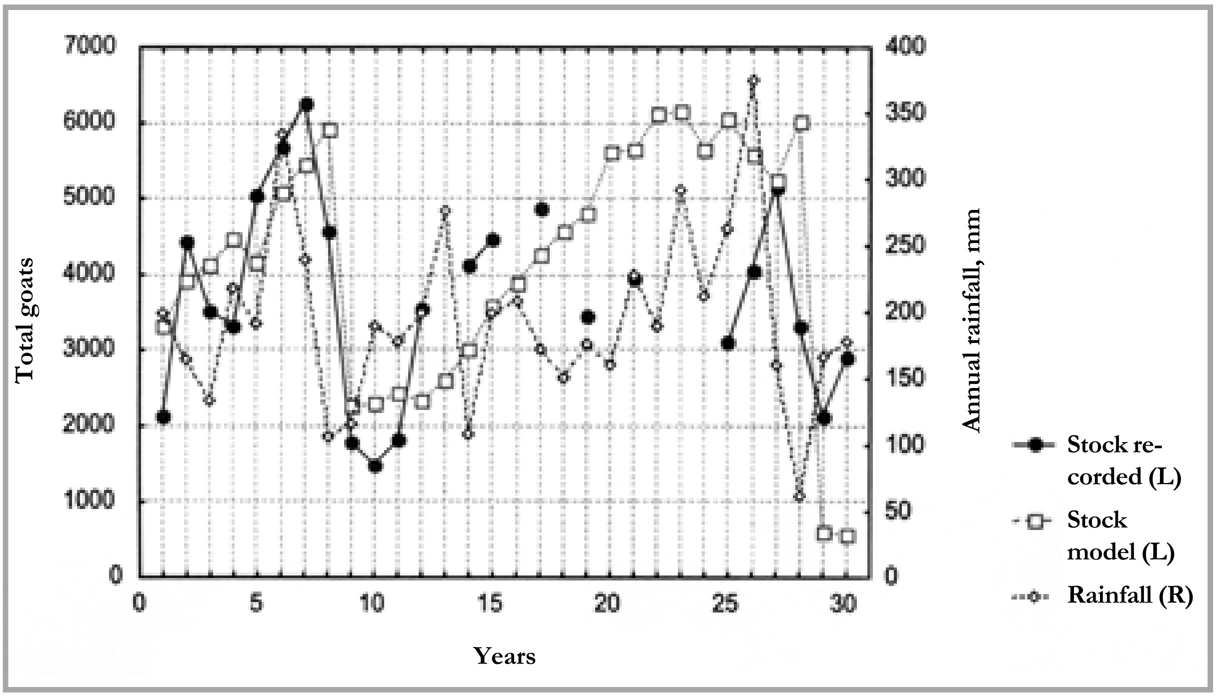


Figure 7: Recorded stock numbers, model predictions and rainfall



Stock numbers over 30 years predicted by the reproductive and mortality segments of the standard long-term model when using recorded rainfall data track recorded stock numbers well (Figure 7).

The model was subsequently run using the 100 years standard rainfall data. Livestock numbers varied widely between years. The percentage cover of each guild also varied between years, but there were no substantial long-term trends (Figure 8) although WP cover initially declined from a relatively high value during the first ten years. This run is the 'standard 100-year run'. Apparently the inter-relations between vegetation, rainfall and stock density are complex.

Figure 8: Variation in stock numbers and cover of three guilds

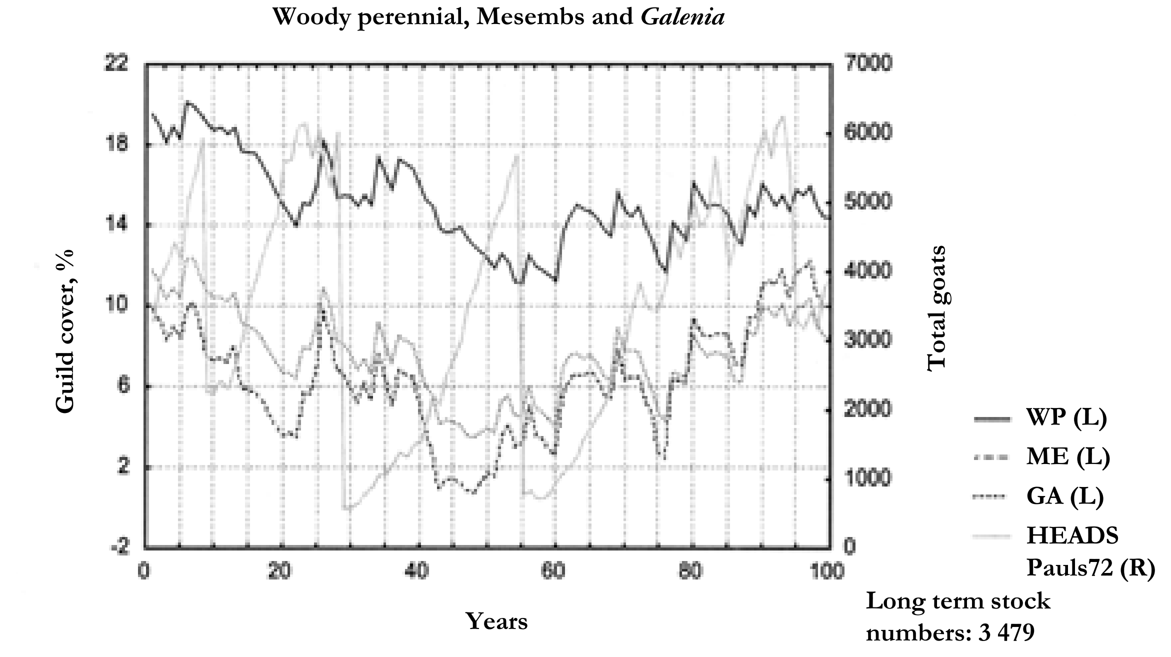
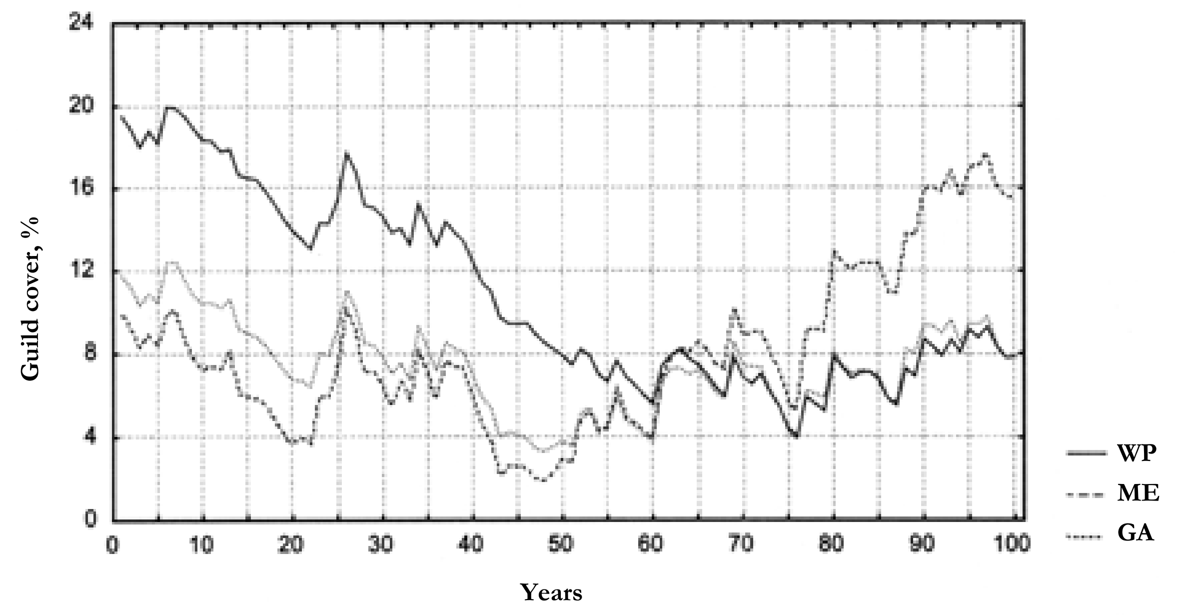


Figure 9: Changes in percentage guild cover, goat numbers held constant at 2 206 SSU



So that different stocking rate strategies can be compared, the number of small stock units (SSU) was estimated for each year of any run. As the young are weaned at six months and then classified as yearlings, the effect of a sucking kid over the whole year is assumed to be 0.25 goat units or SSU. Yearlings are equivalent to 0.8 SSU and adult does are 1 SSU. The 100-year average SSU for the run shown in Figure 8 was 2 206. The model was then run using the same rainfall pattern but livestock numbers were held constant at 2 206 adults. The model predicted that both WP and ME declined progressively and were driven to below 4% by year 75 but recovered during subsequent high rainfall years (Figure 9).

The model was used to examine the effect on vegetation if goat numbers are made to vary at random about the mean and coefficient of variation (CV) in SSU predicted by the standard 100-year run. The actual mean and CV were slightly different from those of the standard run, as 100 years was not long enough to reproduce the variability exactly. The pattern of changes in guild cover was very similar to that predicted when numbers are held constant (Figure 10). Again both WP and ME decline and are virtually driven to extinction by year 60. Apparently the pattern of change in animal numbers over time is the characteristic of the system that prevents a long-term decline in range condition rather than variation in stock numbers per se.

Commercial farmers in the region allow stock numbers to vary between years but limit them to a maximum of 2 000 non-kids (capped at 2 000). The model then predicted that all perennial guilds decline following a sequence of low rainfall years between years 38 and 45. Recovery of vegetation especially WP and ME is predicted from year 61 onwards. This is due to 470mm of rain in year 61 combined with low stock numbers as a result of only 70mm rain in year 57 (Figure 11).

Figure 10: Changes in guild cover and in small stock units

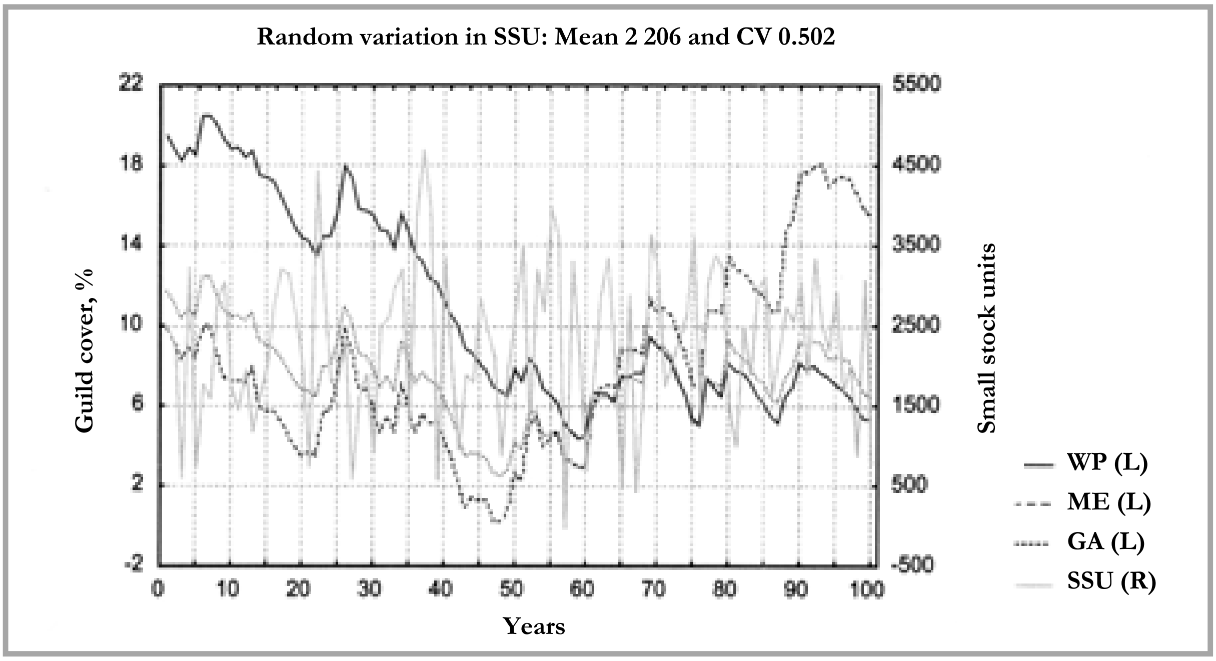
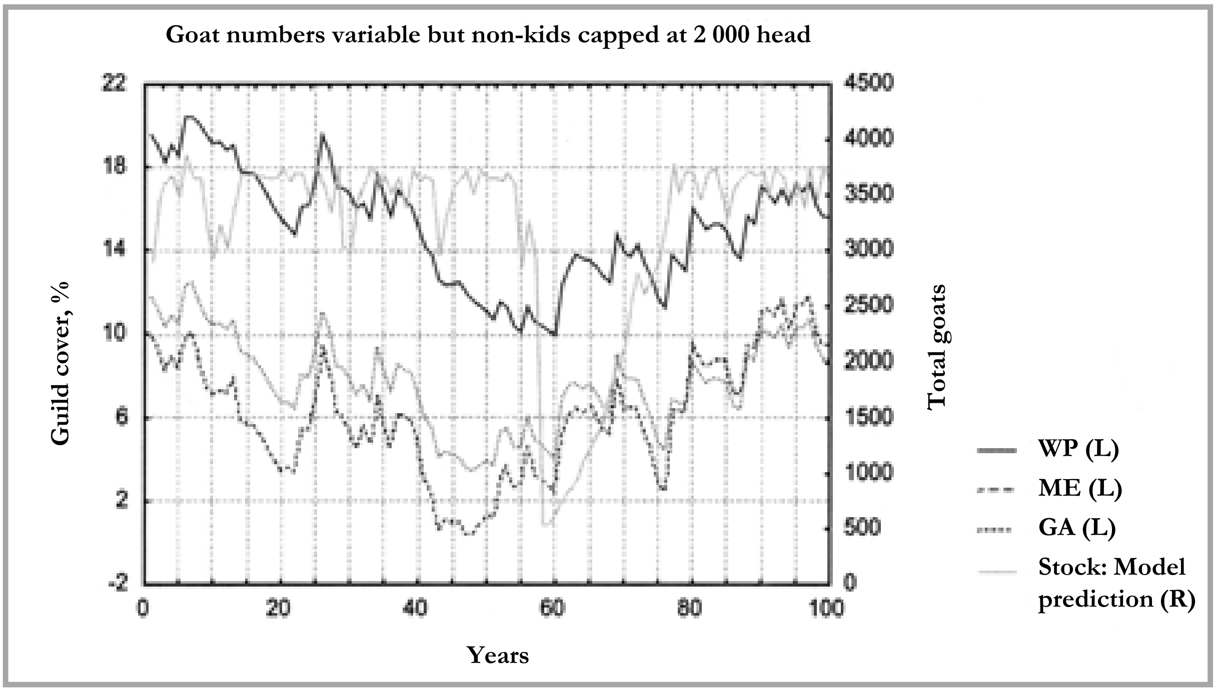


Figure 11: Variation in guild cover and stock numbers



Interestingly, four years of high rainfall from years 77 to 80 lead to an increase in the cover of WP and ME within all stocking regimes, including when there are no grazing livestock (Figure 12).

The mean stocking rate for 20 000 ha over 100 years calculated by the model for the variable numbers strategy is 2 206 SSU and approximately 10% higher than the Department of Agriculture recommendation (Hoffman et al. 1999). For this rainfall pattern there is a marginal benefit in terms of offtake from limiting non-kids to 2 000, due to small increases in reproductive and survival rates (Table 1). Capping at 2 000 non-kids leads to a mean of 2 020 SSU (the recommended rate).

Although further reduction of the maximum number of non-kids to 1 500 leads to another increase in reproductive and survival rates, this is outweighed by a small number of breeding does and offtake is similar to that when no limit is placed on stock numbers (Table 1).

DISCUSSION

SOLUTION TRAJECTORIES OF SIMPLIFIED AND STANDARD MODELS

Differences in the behaviour of the simplified and the standard models using the same rainfall pattern may be studied by plotting their outputs as solution trajectories. These show how the relation between livestock and vegetation changes over time.

The solution trajectory of the output from the simplified model shows that the severe decreases in animal numbers in years 8 and 28 are associated with low rainfall (Figure 13). However unlike the standard long-term model, these specific 'crashes' occur in years when the forage density is also low. The goats increase or maintain

Figure 12: Variation in guild cover over time: no grazing animals

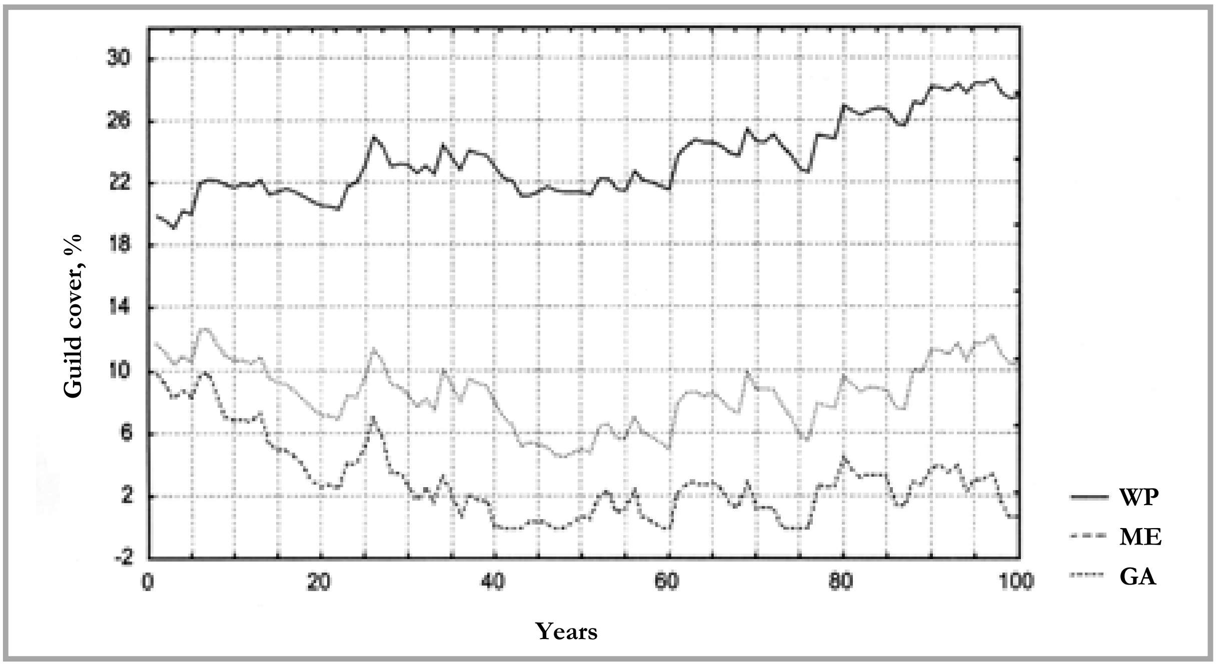
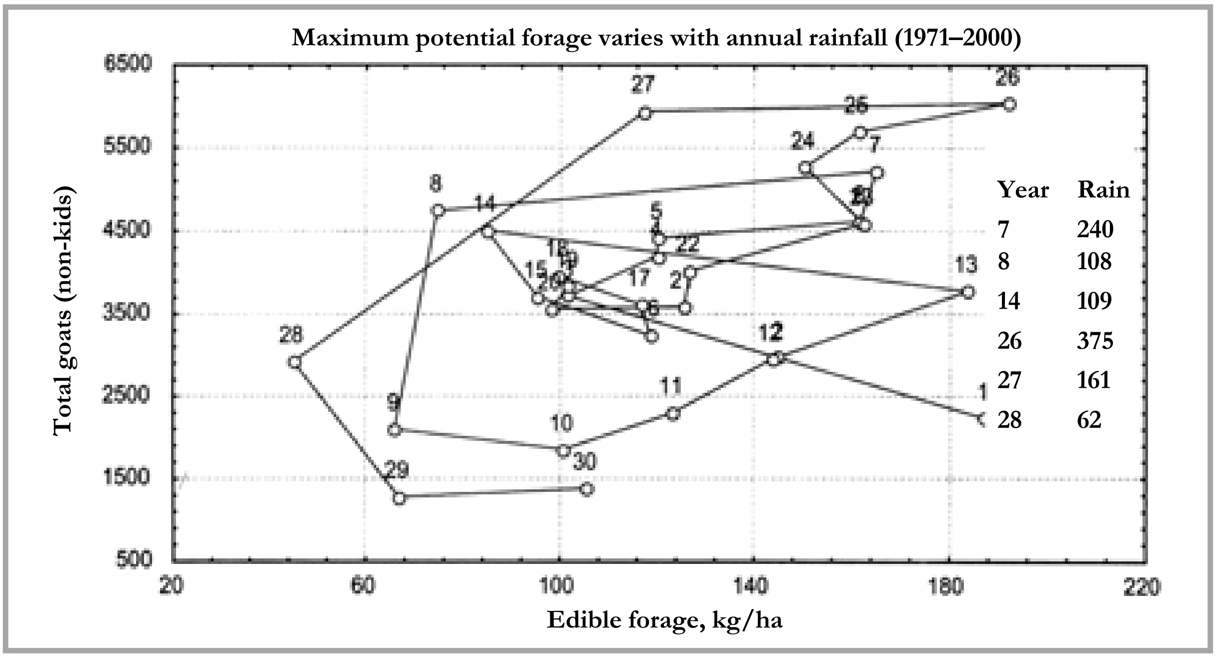


Table 1: Effect of stocking rate strategy on mean stock numbers and offtake and on mean reproductive and survival rates

STOCKING STRATEGY	VARIABLE	CAPPED AT	
		2 000 NON-KIDS	1 500 NON-KIDS
Goat units	2 206	2 020	1 674
Non-kids	1 909	1 747	1 445
Kids	1 570	1 449	1 225
Offtake (sales)	681	754	693
Reproductive rate	1.096	1.103	1.143
Doe survival rate (rearing twins)	0.901	0.925	0.935
Kid survival rate (twins)	0.550	0.576	0.612

Figure 13: Solution trajectory of animal numbers and edible forage



Maximum potential forage varies with annual rainfall (1971–2000)

very high numbers during years of high forage density. High rainfall as in year 26 also enables a large number of animals to be sustained, even though there is a decrease in forage density from years 26 to 27. The behaviour of this model is similar to that shown in Figure 1 of Illius and O'Connor (1999) where random variation in animal mortality prevented the system from reaching equilibrium. Random variation in rainfall and consequently in potential herbage growth simulated by the simplified model also prevented the system from reaching equilibrium. However both the simplified model and that cited by Illius and O'Connor ignore the effect of herbivory on changes in guild composition and its consequent long-term effect on productivity of the vegetation. Furthermore these models also do not take account of the effects of the large proportion of inedible biomass on the inter-relations between vegetation, livestock and rainfall.

For the standard run of the standard model the changes in the relation between WP cover and goat numbers were plotted as a solution trajectory (Figure 14) as WP is the most productive component of the vegetation in terms of edible material. Examination of the solution trajectories supports the concept that the Paulshoek system is driven by rainfall. The severe decreases in animal numbers in years 8, 28 and 54 follow very low rainfall. The crashes in goat numbers apparently occur independently of the number of WP plants/ha, so vegetation condition has either little or no effect on animal population dynamics. There is no rapid increase in WP cover following a crash in the animal population.

The Paulshoek system apparently exhibits non-equilibrium dynamics in respect of the inter-relations between livestock and vegetation. The differences in the behaviour of the two models may be explained by differences in the way the plant components are represented. The standard model partially separates the effects of rainfall and herbivory on the plant population from their influence on the production of edible material for each guild. Furthermore, only 0.15 of the above ground biomass is edible, except for annuals whose biomass is 0.75 edible. The model also simulates each guild separately and they respond in different ways. As *Galenia* is toxic and unpalatable, direct effects of grazing are limited. The increase in cover of this guild, which is associated with a continuous heavy stocking rate, is a consequence of the decrease in competition from palatable guilds such as WP. The growth of annuals is almost entirely dependent on current rainfall.

Survival rates of both does and kids depend on the availability of edible material, of WP and annuals (kg/SSU). Availability is defined as forage density (kg/ha) divided by stocking rate measured as SSU/ha (Table 2). The model predicted that only 0.05 of does and kids in years 28 and 54 would survive when the availability of edible WP plus annuals was less than 310kg/SSU. However, survival rates were unrelated to WP cover. High mortality in year 8 occurred when WP cover was 19.5%, but stock numbers increased in years 21 and 53 when WP cover was 14.5 and 12.2% respectively. Similarly very low reproductive rates (<0.8) invariably occur in the year following a year of low rainfall (<150mm) (Figure 15). If a different pattern of rainfall having the same mean and variability is used, the system still responds to rainfall in a similar way.

Changes in the number of WP plants per ha are the result of interactions between rainfall, amounts of WP eaten

Figure 14: Solution trajectory of goat numbers in relation to WP cover: Variable livestock numbers: years 1 to 60

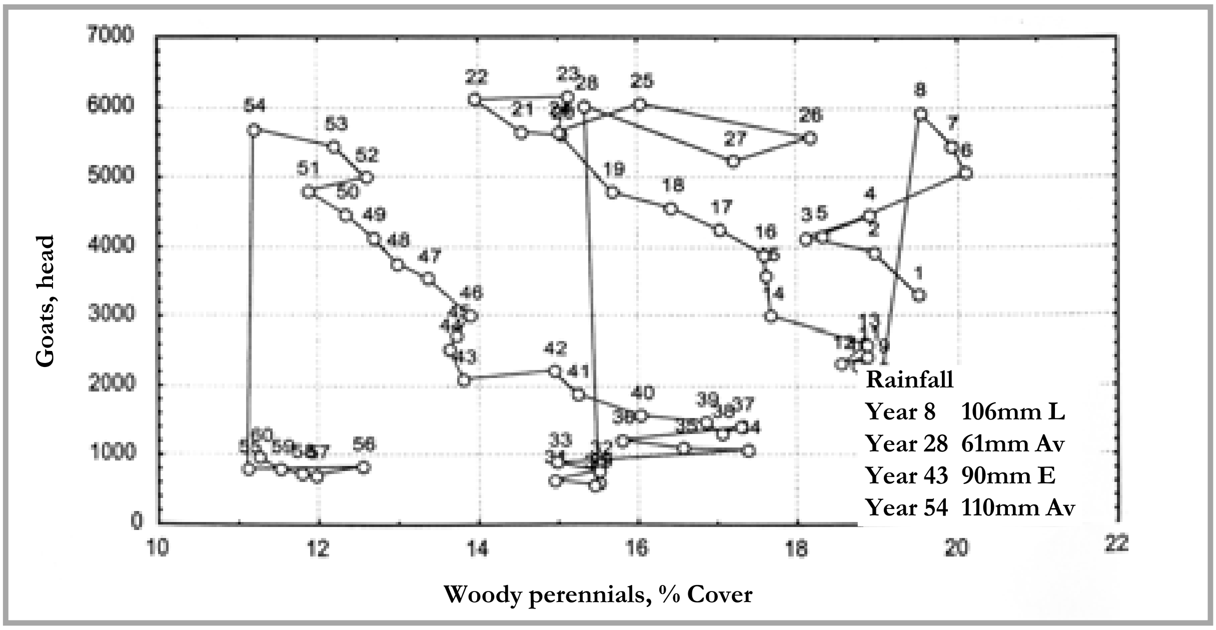


Figure 15: Inter-annual variation in rainfall and reproductive rate

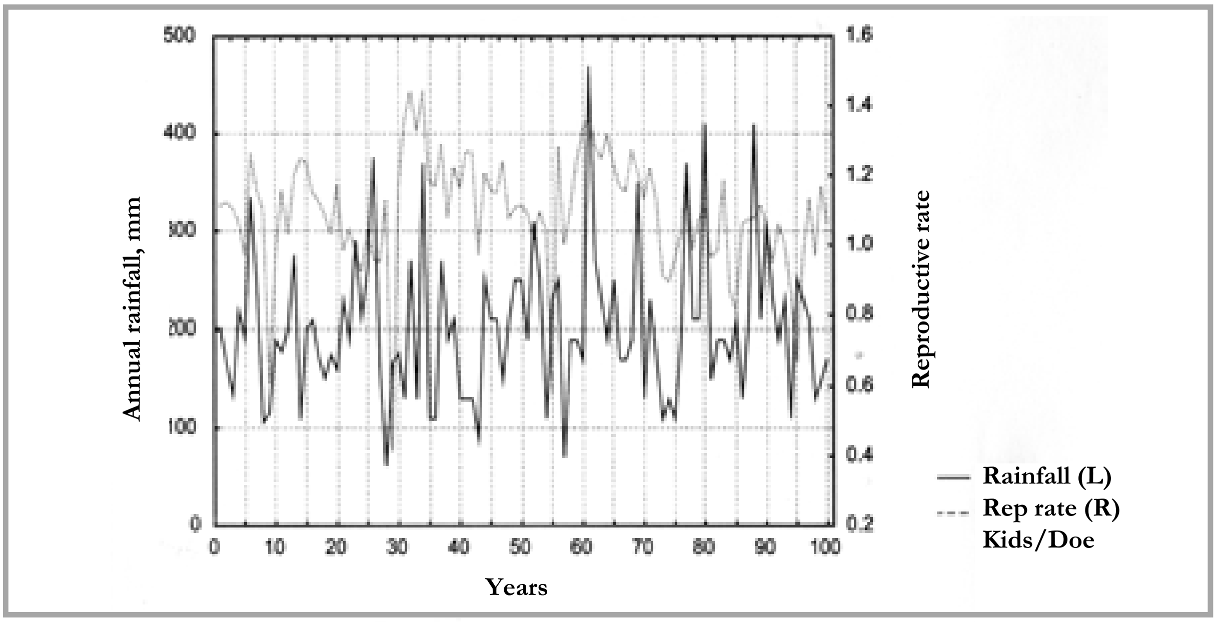


Table 2: Forage availability and goat survival in years of different rainfall

YEAR	RAIN	SSU	EDIBLE FORAGE				SURVIVAL RATES	
			KG/HA		KG/SSU		SINGLE KIDS	
			WP	An	WP	An	Does	Kids
7	240 Av	3149	181	47	1149	298	0.95	0.80
8	106 L	3436	69	27	402	157	0.48	0.50
27	161 E	3100	99	12	639	77	0.96	0.88
28	62 Av	3453	35	0	203	0	0.05	0.05
53	250 Av	3130	117	37	748	236	0.95	0.78
54	110 Av	3350	42	9	251	54	0.05	0.05

and competition with other guilds (Figure 16). Edible WP density depends on rainfall, the number of WP plants per ha and grazing.

Comparing the Paulshoek models with other grazing ecosystems models may increase our understanding of the dynamics of grazing systems. Output from the model of a semi-arid savanna ecosystem developed by Hahn et al. (1999) indicates that severe decreases in animal numbers are associated with a decrease in grass yield following a period when animal numbers are high and rainfall is low. Thus the savanna system would tend towards equilibrium, as suggested by Illius and O'Connor (1999).

Differences in the structure of the simulated forage plants may explain the difference in behaviour between the model of a dwarf shrub pastoral system and that of the savanna grazing system. In the Paulshoek model only 0.15% of the shrub biomass is assumed to be edible and although 0.75% of biomass of annuals is edible it forms only a small part of the total vegetation. On the other hand the savanna model assumes that virtually the whole of the grass biomass is edible. The standard model assumes that changes in the shrub population and the production of edible forage by shrubs are partially independent of one another, whereas in the savanna model there is only one variable representing grass growth. These emphasise the need to refine and expand our knowledge of individual rangeland systems as suggested by Sullivan and Rohde (2002).

CONCLUSIONS

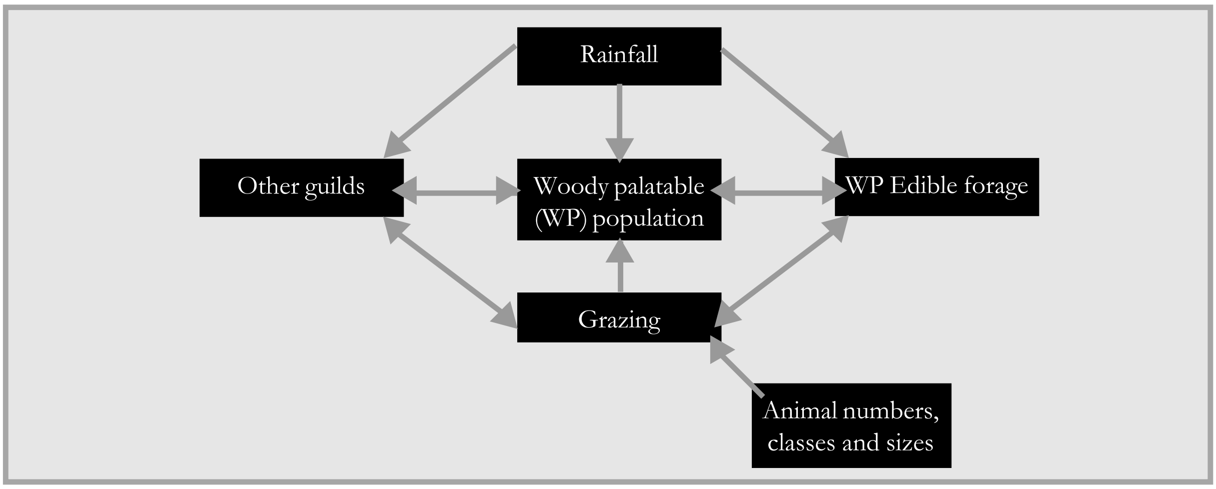
The simplified model does not realistically simulate the inter-relations between rainfall, vegetation and livestock as the complex nature of vegetation is ignored. The diversity of plant guilds needs to be simulated, as does the partial independence of edible material production from changes in the population of each perennial guild. This has been done in the standard model and as a result predicted livestock numbers are similar to the recorded numbers for most years. The 100-year mean stock numbers of 2 206 (0.1 103 SSU/ha) predicted by the model (Table 1) when no measures are taken to limit livestock or to prevent mortality or to import animals after a drought may be termed the long-term *ecological* carrying capacity (Behnke & Scoones 1993). Mean offtake measured as animals sold and slaughtered is apparently maximal when the number of non-kids is limited to 2 000 and the long-term average SSU is 2 020 which may be considered as the long-term *economic* carrying capacity and is almost identical to the recommendation of the Department of Agriculture. However these terms are in a way meaningless as the range is unable to support either of these stock populations during years of low rainfall. The long-term average small stock numbers reported for the area by Hoffman et al. (1999) are larger than the numbers predicted by the model and may indicate that numbers could have been boosted following droughts by the importation of additional animals.

Maintenance and recovery of the desirable WP apparently occurs as a consequence of the sharp decline and slow recovery of stock numbers following a drought. The temporal pattern of variation in rainfall and animal numbers combined with characteristics of the vegetation is the mechanism determining the changes in guild populations. Neither the concept of equilibrium nor that of non-equilibrium is adequate to explain the behaviour of the system. The present model does not simulate the effects of key resources, if any, on the performance of the system as a whole.

REFERENCES

Behnke, R & Scoones, I. 1993. Rethinking range ecology: Implications for rangeland management in Africa, in *Range Ecology at disequilibrium: New models of natural variability and pastoral adaptation in African savannas*, edited by R Behnke, I Scoones & C Kerven. London:

Figure 16: Woody palatable shrubs sub-model



Overseas Development Institute.

Briske, D, Fuhlendorf, S & Smeins, F. 2003. Vegetation dynamics on rangelands: A critique of the current paradigms. *Journal of Applied Ecology*, 40:601–14.

Crawley, M. 1983. *Herbivory: The dynamics of animal–plant interactions.* Oxford: Blackwell Scientific.

Ellis, J. & Swift, D. 1988. Stability of African pastoral ecosystems: Alternate paradigms and implications for development. *Journal of Range Management,* 41:450–9.

Hahn, BD, Richardson, FD & Starfield, A. 1999. Frame-based modelling as a method of simulating rangeland production systems in the long term. *Agricultural Systems,* 62:29–99.

Hahn BD, Richardson, FD, Hoffman, MT, Roberts, R, Todd, S & Carrick, P. 2003. A simulation model of long-term climate, livestock and vegetation interactions on communal rangelands in the semi-arid succulent Karoo, Namaqualand, South Africa. Submitted to *Ecological Modelling*.

Hoffman, MT, Cousins, B, Meyer, T, Petersen, A & Hendricks, H. 1999. Historical and contemporary land use and the desertification of the Karoo, in *The Karoo: Ecological patterns and processes*, edited by W Dean & S Milton. Cambridge: Cambridge University Press.

Illius, A & O'Connor, T. 1999. The relevance of non-equilibrium concepts to arid and semi-arid grazing systems, in *Ecological Applications,* 9:798–813.

Richardson, FD, Hahn, BD & Schoeman, S. 2000. Modelling nutrient utilization by livestock grazing semi-arid rangeland. In *Modelling nutrient utilization in farm animals*, edited by J McNamara, J France & D Beever. Wallingford, Oxfordshire: CABI.

Sullivan, S & Rohde, R. 2002. On non-equilibrium in arid and semi-arid grazing systems. *Journal of Biogeography*, 29:1595–618.

Todd, S & Hoffman, MT. 1999. A fence line contrast reveals effects of heavy grazing on plant species diversity and community composition in Namaqualand, South Africa. *Plant Ecology*, 142:169–78.

EQUILIBRIUM AND NON-EQUILIBRIUM BEHAVIOURS OF RANGE VEGETATION DYNAMICS IN THE SAHEL

PIERRE HIERNAUX

INTRODUCTION

Because of the wide fluctuations in annual rainfall and the unpredictable rainfall pattern within rainy seasons, the Sahel ecosystems are considered good candidates for non-equilibrium dynamics behaviour (Behnke & Scoones 1993). Sahelian rangelands are even considered ‘non-equilibrium ecosystems’, meaning that changes in vegetation should verify the non-equilibrium behaviour model and vegetation states should all be transient (Ellis 1994). These hypotheses are further supported by the leading role of pastoral production systems in natural resource management with plurality of animal species, mobility of livestock and communal land tenure resulting in patchy and variable grazing pressures. However, no formal validation from field survey, monitoring and experimental data has been conducted so far in the Sahel (Thébaud et al. 1995; Rietkerk et al. 1996). This paper examines data from rangeland surveys and monitoring, and from grazing experiments carried out in the Sahel and discusses their use in validating equilibrium or non-equilibrium behaviour models. Implications for policies and management are drawn in conclusion.

THE DOUBLE DUALISMS:

EQUILIBRIUM AND NON-EQUILIBRIUM BEHAVIOUR, CLIMAX SUCCESSION AND STATE-AND-TRANSITION MODELS OF VEGETATION DYNAMICS

The concept of ‘non-equilibrium’ in vegetation dynamics is first defined by opposition to the ‘equilibrium’ concept that supports the climax succession model of vegetation dynamics (Clements 1916; Clements 1936; Smith 1988). Indeed, the climax is defined as an ecosystem steady-state, or ‘dynamic equilibrium’, in which the use of environmental resources for sustainable biological production is maximised (Whittaker 1953). This is achieved by reaching maxima in biomass, diversity and complexity of the trophic system while productivity is often inferior to what it is in mid-succession states (Odum 1971). The climax succession model is a linear sequence of vegetation states stretching from an abiotic mineral state followed by pioneer states, and a series of ‘seral’ states that lead to the final ‘climatic climax’. The changes from one state to the next are gradual and driven by competition between plants (McCook 1994) and plant-herbivore interactions (Chesson & Case 1986). Changes are also reversible in that they can either upgrade the vegetation toward the climax, or retrograde the vegetation away from the climax. The linearity is however less exact when the succession is interrupted by a major disturbance that results in a transient state, away from the succession but from which a secondary succession catches up with the succession (homeorhesis) toward the same ‘climatic climax’ (Tueller 1973). When the disturbance repeats itself, such as by regular burning, the disturbed state constitutes a sort of secondary state of equilibrium, named either ‘para-, pseudo-, peni-, or disclimax’ (Trochain 1940), or ‘pyroclimax’ when maintained by burning (Roberty 1946). However, this secondary climax is considered unstable and supposed to revert to the primary succession as soon as the constraint is removed.

Although under most climate and edaphic conditions rangelands would be considered early stages of a climax succession that would finally lead to a forest climatic-climax, the succession model has been applied separately to the rangeland segment of the succession. In this application, the climax is represented by a rangeland in ‘excellent condition’ (Dyskterhuis 1949) in which herbaceous standing mass is maximal and deep-rooted perennial grasses dominate a species-rich herbaceous community (Holechek et al. 1989). Grazing over a threshold of intensity (overgrazing) leads to gradual changes of flora composition, species being classified into ‘invaders’, ‘increasers’ and ‘decreasers’ depending on their density trend when the range is subjected to increasing grazing pressure. Because of the linearity of the model the changes in flora with increasing grazing pressure are supposed to be systematically accompanied

by losses in productivity, forage value, stability and resilience to environment stress.

By contrast, 'non-equilibrium' vegetation behaviours are associated with the empirical 'states-and-transition' models that were proposed to remedy the inability of the climax succession model to explain vegetation dynamics events such as severe and apparently irreversible plant invasion, or the existence of alternative and quite stable states (Whittaker 1953; Mentis et al. 1989; Westoby et al. 1989). In state-and-transition models the arrangement of vegetation states is not linear and may bifurcate into alternative states. The transitions between states are not systematically reversible (Walker 1986; Ellis & Swift 1988; Tausch et al. 1993). Some of the state changes are abrupt, and for that reason, are not qualified as 'transitions' but 'instabilities' or 'jumps'. They characterise 'non-equilibrium' behaviour and are associated either with 'non-linear feedback and time lags' or 'hysteresis' in interactions between species (Loehle 1989), or with stochastic forcing in a fluctuating environment, also named 'pulses' (DeAngelis & Waterhouse 1987; Walker 1988). At the extreme, stochastic forcing and non-linear feedbacks are such that they impede an equilibrium state from being reached and render all vegetation states transient.

In the application of the non-equilibrium behaviour to rangeland dynamics, stochastic forcing has been associated with abiotic factors such as climate events, or floods, as opposed to biotic variables such as grazing which are considered to apply more gradually (Scoones 1995). And it is proposed to differentiate 'non-equilibrium' prone ecosystems from 'equilibrium' prone ecosystems by the degree of variation of the abiotic environment (Ellis & Swift 1988). However, grazing pressure is also recognized as a potential source of non-equilibrium behaviour in relation to the decoupling of the herbivores and the range resources (Illius & O'Connor 1999). Indeed in these cases, the large temporal fluctuation in forage resource availability impedes the establishment of equilibrium between herbivores and range at any point in time. Catastrophe mathematical models have been adapted to simulate the dynamic behaviour of non-equilibrium systems, determining the environment conditions and thresholds under which 'jumps' and 'hysteresis' occur (Loehle 1989; Rietkerk & Van de Koppel 1997). The three-dimensional graph of this model (Rietkerk et al. 1996) indicates that gradual and oriented changes take place at any point of surface that represent environment conditions away from the catastrophe manifold area where, depending on the direction of change, either jumps or hysteresis will occur. Gradual changes and jumps appear, just as the two extremes on the range of vegetation dynamics behaviour (Wiens 1984). Thus the model does not imply that there are distinct non-equilibrium and equilibrium ecosystems, but that non-equilibrium and equilibrium behaviours may occur in the same ecosystem depending on environmental circumstances (Huston 1979; Briske et al. 2003). The model does not preclude either a biotic or abiotic nature of the driving variables that determine equilibrium or non-equilibrium type of behaviour. Grazing could trigger vegetation jumps or contribute to hysteresis, as well as fire, climatic or other abiotic events. Gradual changes and abrupt jumps contribute to transitions between vegetation states within a broader 'state-and-transition' model (for example, Hiernaux 2000; 2001 for the herbaceous vegetation of the Sahel) and relate*, in fine,* to the biological mechanisms that drive species populations dynamics: seed production and dispersal, seed dormancy and germination and competitive growth.

LESSONS FROM VEGETATION SURVEYS

As in other arid and semi-arid ecosystems, the vegetation pattern in the Sahel is structured in a hierarchy of embedded scales (Friedel 1994). At the lower levels of that scale, competition between individual plants for water, nutrients and solar energy is the structuring process (Tilman 1990), while flows, sources and sinks of water, nutrients, but also propagules, herbivores and human activities structure the elements at higher levels of the hierarchy (Walker 1986). In the Sahel, vegetation is composed of two main plant functional groups: the herbaceous annuals and the woody plants. A few herbaceous perennials can also contribute locally (or at one point over time) to the rangeland ecosystem. Micro-heterogeneity has often been recognized as a characteristic trait of the annual herbaceous layer of sahelian rangelands (Bille 1977; Grouzis 1988; Hiernaux 1996). However, the distribution patterns of annuals and woody plants are organised at different scales. Indeed the influence of individual woody plants is one of the structuring variables of herbaceous patchiness whose unit area typically ranges between 50 and 250m^2 in arid as well as more humid west African savannas (Poisonnet & César 1972; Fournier 1991; Hiernaux 1998). Other local edaphic variables explain some of more permanent patches matching with the spatial extent of large termite mounds, ant nests, squirrel and jackal burrows, and micro-topography units (Brouwer et al. 1993; Tongway & Ludwig 1994). Other patches, however, are not related to obvious edaphic nuances and may change from year to year in location and shape. This later type of patchiness tends to decrease as grazing pressure increases (Hiernaux 1998). In the Sahel, the high degree of patchiness of the herbaceous layer does not correspond to narrow ecological specialisation of species. Most species have quite wide ecological niches, and grouping of species that characterise patches in one site and year are unstable. When there are no differences in land use, larger size structures such as herbaceous 'facies' and woody plant communities are structured by water and nutrient distribution often related to topography, geomorphology

and soils (Breman & de Ridder 1991). Current land use pattern and its history overlay that of natural vegetation, either masking or enhancing its pattern, depending on their intensity and spatial organisation.

LESSONS FROM VEGETATION MONITORING

High inter-annual variation in species composition of the herbaceous layer has been often observed in the Sahel (Breman & Cissé 1977; Cissé 1986; Grouzis 1988; Miehe 1998; Hiernaux 1996). The woody plant populations change less rapidly, although spectacular dying off or recruitment events have also been observed (Boudet 1972; Couteron & Kokou 1997). The hypothesis that vegetation changes are moderated by the heterogeneity of vegetation pattern has not been verified (Hiernaux 1996). Seed production, dispersal, conservation and germination are determinant in annual dynamics, as seed dormancy in most species is seasonal. High rates of germination are observed with the first rains as an adaptation to the regular seasonality of the tropical wet season (Cissé 1986). The annual seed stock is thus largely transient, with only a few species with either staggered or delayed germination (Carrière 1989; Herault & Hiernaux 2003). These species are favoured by dry spells or more severe droughts that reduce or suppress the competition from the dominant early germinating species, causing spectacular shifts in herbaceous vegetation composition (Breman & Cissé 1977). Seed production and dispersal is also determinant to the recruitment of woody or herbaceous perennials, which often occurs in cohorts triggered by exceptional events: large rains, flood, burning, droughts, heavy browsing by elephants, clearing by man, locust or bird infestations. The dormancy of perennial plant seeds is often extended, with risks of high seed losses explaining the low density of seedlings in spite of large seed production. Longevity of the established plants and vegetative reproduction are thus determinant to the overall dynamics of the perennial plant population.

Large and abrupt changes of the vegetation in response to changing abiotic variables do not prevent vegetation from responding to grazing when grazing affects one of the processes of the plant reproductive cycle. Trampling can affect seed dormancy and the germination process, selective grazing can influence the competition between species, livestock can contribute to seed dispersal of particular species in specific patterns, and lastly, the uneven distribution of excretions in the landscape creates patches of higher soil fertility that may influence vegetation dynamics (Turner 1998). The influence of grazing can also reinforce other factors such as burning or drought. The contrast between the short wet and the long dry season in the Sahel is a base for the decoupling of livestock and rangelands. Indeed, if animals are not supplemented or supplementation is limited to a small fraction of the diet, the survival of livestock during the dry season depends on access to key resources during the late part of that season and early wet season (Scoones 1995). This is often controlled by access to water (Thébaud 1990; Coughenour 1991). As a consequence, there should be no or weak functional relationships between the density of herbivores and the herbage growth during the wet season, precisely when trampling and defoliation could have a large effect on soil surface and vegetation growth (Hiernaux et al. 1999). Decoupling livestock and overall range resources, however, does not always mitigate the effect of livestock grazing on rangeland as it generates extreme grazing pressures on key resources. Livestock mobility aims at optimising the use of pastoral resources by grazing animals through ‘opportunistic’ tracking strategies, which remain individualistic despite elaborated social institutions (Bonfiglioli 1988). Mobility is key to adjusting grazing pressure to available forages, but it does not prevent local concentration of grazing pressure over key forage resources (Schlecht et al. 2001).

LESSONS FROM GRAZING AND EXCLOSURE EXPERIMENTS

In the Sahel, controlled grazing experiments have been conducted in fenced paddocks, and attempted to measure the effects of different stocking rates under continuous grazing with a single animal species (Guerin 1988; Ickowicz 1995; Thébaud et al. 1995; Ayantunde et al. 1999). A few experiments have demonstrated advantages of a species mix over single species herds in animal performance and grazing impact on the rangeland (Nolan et al. 1989; Hiernaux & Fernández-Rivera 1995). Experimental results are not easily transferable to communal grazing in open rangelands with a mix of animal species and changing stocking rates; however they reveal a flaw of the carrying capacity and succession concepts applied to rangelands, which predict the effects of grazing on herbage (cover, production, composition, resilience) and soil (porosity, water retention, erosion) to be systematically detrimental and proportional to the stocking rate. There is experimental evidence in the Sahel that contradicts these statements. Grazing seldom results in mortality of established plants following severe defoliation, either because of the short growing cycle (annuals) or because of specific plant-defences (woody plants), but it affects production. Although it has been much debated (Belsky 1986; Milchunas & Lauenroth 1993), compensatory growth of grazed plants has been measured in tropical rangelands (Oesterheld & McNaughton 1991), including in the Sahel with annual plants (Hiernaux & Turner 1996) and browses (Oba, 1998).

Increases in feed value of grazed rangelands have either been attributed to the short-term rejuvenation of plant organs, or to the longer term changes in species composition (Cissé 1986; Miehe 1998). The richness of the herbaceous flora generally increases with protection from grazing (Thébaud et al. 1995; Hiernaux &

Fernández-Rivera 1999). However, in a fallow part of semi-arid Niger, the increase was larger after three years of protection than after 14 years. Moderate grazing maintained flora richness through fluctuating rainfall conditions and heavy continuous grazing decreased flora richness, but the flora was more affected after 2–3 years of heavy grazing by sheep than by 8–9 years of intense mixed grazing (Hiernaux 1998). Trampling, selective defoliation and deposition of faeces and urine by grazing livestock affect seed dispersal, germination and early development of seeds (Cissé 1986; Carrière 1989; O'Connor & Pickett 1992). When grazing events occur at a critical time of plant development, in the early wet season, they may trigger abrupt changes in the vegetation even at moderate stocking rate (Breman & Cissé 1977; Hiernaux & Fernández-Rivera 1999; Hiernaux & Hérault 2003).

The patchiness of the herbaceous layer was enhanced by long-term protection resulting in a mosaic of highly contrasted but internally homogeneous patches (Hiernaux 1998). At moderate grazing pressure, the rather flat distribution of species over space indicated a more aggregated arrangement of less differentiated but internally diverse patches. Heavy grazing led to the dominance of a few species but also left niches for scattered individuals of other species. A majority of species encountered in old Sahelian fallows were either favoured by, indifferent to, or tolerant of grazing (Gérard et al. 2001). However, more than a third of the species appeared sensitive to heavy grazing, and no relationships were found between species response to grazing and palatability or productivity. The list of species recorded under different grazing or protection treatments did not differ significantly with regard to the proportion of perennials, legumes, species palatable to ruminants, or species whose seeds are dispersed by mammals. However, the number of grass species, short cycle annuals and xerophytes increased with grazing intensity (Hiernaux 1998). The contrasted effects ('fence lines') that the same stocking rates have had on the rangeland depending on animal species (Hiernaux & Fernández-Rivera 1995) and on the timing of grazing (Hiernaux & Fernández-Rivera 1999), demonstrated that stocking rate alone does not explain the impact of grazing.

CONCLUSION

The opportunistic resource tracking strategies and regional mobility of herds have demonstrated their efficacy in enhancing herd production and limiting risks (De Ridder & Wagenaar 1984; Thébault 1990). However, opportunistic management strategies do not prevent rangeland degradation, and contribute to the maintenance of an inequity between crop and pastoral land tenure status (Bonfiglioli 1990). This inequity facilitated the expansion of cropping at the pace of the population growth and to the detriment of the area of rangelands progressively confined to drier zones and marginal lands. As a consequence, the grazing pressure exerted on the remaining rangelands is increased, especially during the wet season when livestock are kept away from the cropped lands, and when rangelands are most sensitive to grazing (Ayoub 1998).

The short term effects of controlled rotation and deferred grazing on herd production and herbage yields are slight and ambiguous, but their long-term effects on the species composition and soil surface features of the range can be substantial, especially in the semi-arid belt where the land remains uncropped, and rangelands are subject to continuous high grazing pressure (Hiernaux 1998). However, controlled grazing, even when based on the intensity and timing of grazing in relation to plant phenology rather than stocking rates, does not cope with the fluctuations of forage resources over space and time due to the climate vagaries (Thébaud et al. 1995). A compromise solution would consist of promoting the regional mobility of livestock and the control of local grazing under communal institutions. Livestock mobility is organized at two embedded scales in time and space: livestock move locally through daily grazing itineraries, and regionally through seasonal transhumance and nomadic movements, including exceptional migrations in case of drought, flood, epizooty or war outbreak (Turner 1998). Collective action at the local level, relayed by corporate and political interventions at the regional level, should promote and reinforce the specialised institutions which are required to strengthen the capacities of pastoralists' associations (Vedeld 1992; Marty 1993). Controlled grazing organised locally requires active herding and collective action to reach agreement within a community and with the other partners on the mode and calendar of rangeland use. Specialised institutions are needed to implement these tasks. During that process, the recognition by neighbouring communities and by administration of the prerogatives of the community to manage the pastoral resources of the demarcated lands could be perceived as a major advantage by the community and facilitate the promotion of that policy (Niamir-Fuller 1999).

REFERENCES

Ayantunde, A, Hiernaux, P, Fernandez-Rivera, S, Van Keulen, H & Udo, H. 1999. Selective grazing by cattle on spatially and seasonally heterogeneous rangelands in the Sahel. *Journal of Arid Environments*, 42:261–79.

Ayoub, A. 1998. Extent, severity and causative factors of land degradation in the Sudan. *Journal of Arid Environments*, 38:397–409.

Behnke, R & Scoones, I. 1993. Rethinking range ecology: Implications for range and livestock development in Africa, in *Range ecology at disequilibrium: New models of natural variability and pastoral adaptation in African savannas*, edited by R Behnke, I Scoones & C Kerven. London: Overseas Development Institute.

Belsky, A. 1986. Does herbivory benefit plants? A review of evidence. *The American Naturalist*, 127: 871–92.

Bille, J. 1977. Etude de la production primaire nette d'un écosystème sahélien. *Collection Travaux et Documents*, 65. Paris: Orstom.

Bonfiglioli, A. 1988. Dudal. Histoire de famille et histoire du troupeau chez un groupe de Wodaabé du Niger. *Edition de la Maison des Sciences de l'Homme*, Paris: Cambridge University Press.

Bonfiglioli, A. 1990. Pastoralisme, agropastoralisme et retour: itinéraires sahéliens. *Cah. Sci. Hum.* 26(1–2): 255–66.

Boudet, G. 1972. Désertification de l'Afrique tropicale sèche. *Adansonia*, ser 2, 12(4):505–24.

Breman, H & Cissé M. 1977. Dynamics of sahelian pastures in relation to drought and grazing. *Oecologia*, 28(4):301–15.

Breman, H & De Ridder, N. 1991. Manuel sur les pâturages des pays sahéliens, Paris: Karthala.

Briske D, Fuhlendorf S & Smeins, F. 2003. Vegetation dynamics on rangelands: A critique of the current paradigms. *Journal of Applied Ecology*, 40:601–14.

Brouwer, J, Fussell, L & Herrmann, L. 1993. Soil and crop growth micro-variability in the West African semi-arid tropics: A possible risk-reducing factor for subsistence farmers. *Agriculture Ecosystems and Environment,* 45: 229–38.

Carrière, M. 1989. Les communautés végétales sahéliennes en Mauritanie (Région de Kaédi); analyse de la reconstitution annuelle du couvert herbacé. Maisons-Alfort, (France): IEMVT-CIRAD.

Chesson, P & Case, T. 1986. Overview: Non-equilibrium community theories: Chance, variability, history and coexistence, in *Community ecology* edited by J Diamond & T Case. New York: Harper and Row:229–39.

Cissé, A. 1986. *Dynamique de la strate herbacée des pâturages de la zone sud-sahélienne.* Unpublished thesis, Wageningen Agriculture University.

Clements, F. 1916. *Plant succession: An analysis of the development of vegetation*. Washington, DC: Carnegie Institution.

Clements, F. 1936. Nature and the structure of the climax. *Journal of Ecology*, 24:252–284.

Coughenour, M. 1991. Spatial components of plant-herbivore interactions in pastoral, ranching, and native ungulate ecosystems. *Journal of Range Management*, 44:530–42.

Couteron, P & Kokou, K. 1997. Woody vegetation spatial patterns in a semi-arid savanna of Burkina Faso, West Africa. *Plant Ecology*, 132:211–27.

DeAngelis, D & Waterhouse, J. 1987. Equilibrium and non-equilibrium concepts in ecological models. *Ecological Monographs*, 57(1):1–21.

De Ridder N & Wagenaar, K. 1984. A comparison between the productivity of traditional livestock systems and ranching in eastern Botswana. *ILCA Newsletter* 33:5–7.

Dyksterhuis, E. 1949. Condition and management of rangeland based on quantitative ecology. *Journal of Range Management*, 2(3):104–15.

Ellis, J. 1994. Climate variability and complex ecosystems and dynamics: Implication for development in *Living with uncertainty: New directions in pastoral development in Africa*, edited by I Scoones. London: Intermediate Technology:37–46.

Ellis, J & Swift, D.1988. Stability of pastoral ecosystems: Alternate paradigms and implications for development. *Journal of Range Management*, 41:450–9.

Fournier, A. 1991. *Phénologie, croissance et production végétale dans quelques savanes d'Afrique de l'Ouest, variation selon un gradient climatique*. Paris: Orstom. (Collection etudes et thèses.)

Friedel, M. 1994. How spatial and temporal scale affect the perception of change in rangelands. *Rangeland Journal,* 16:16–25.

Gérard B, Hiernaux, P, Muehlig-Versen, B & Buerkert A. 2001. Destructive and non-destructive measurements of residual crop residue and phosphorus effects on growth and composition of herbaceous fallow species in the Sahel. *Plant and Soil*, 228:265–73.

Grouzis, M. 1988. *Structure, productivité et dynamique des systèmes écologiques sahéliens (mare d'Oursi, Burkina Faso).* Paris: Orstom. (Collection etudes et thèses.)

Guerin, H. 1988. *Alimentation des ruminants domestiques sur pâturages naturels sahéliens et sahélo-soudaniens; étude méthodologique dans la région du Ferlo au Sénégal*. Thèse, Université des Sciences et Techniques du Languedoc. Montpellier: IEMVT-CIRAD.

Hérault, B & Hiernaux, P. 2003. Dynamics of annual rangelands in the Sahel: Soil, seed stock and germination pattern, in *Proceedings of the VIIth International Rangelands Congress, Durban, South Africa*, edited by N Allsopp, A Palmer, S Milton, K Kirkman, G Kerley, C Hurt & C Brown:356–8.

Hiernaux, P. 1996. Spatial heterogeneity in Sahelian rangelands and resilience to drought and grazing, in *Rangelands in a sustainable biosphere: Proceedings of the Vth International Rangelands Congress*, vol 2, edited by N West. Denver, Colorado: Society for Range Management:232–3.

Hiernaux, P. 1998. Effects of grazing on plant species composition and spatial distribution in rangelands of the Sahel. *Plant Ecology*, 138:191–202.

Hiernaux, P. 2000. Implications of the 'new rangeland paradigm' for natural resource management, in *The Sahel: Energy supply, economic pillars of rural Sahelian communities, need for revised development strategies*, edited by H Adriansen, A Reenberg & I Nielsen. Proceedings of the XIIth Danish Sahel Workshop. Copenhagen:

Sudan-Sahel Environmental Research Initiative (SEREIN):113–42. (Occasional paper; no. 11.)

Hiernaux, P. 2001. Fondements ecologiques de la gestion des parcours au Sahel, in *Elevage et gestion de parcours au Sahel: Implications pour le developpement*, edited by E Tielkes, E Schlecht & P Hiernaux. Stuttgart: Verlag E Grauer:65–77.

Hiernaux, P, Bielders, C, Valentin, C, Bationo, A & Fernández-Rivera, S.1999. Effect of livestock grazing on physical and chemical properties of sandy soils in Sahelian rangelands. *Journal of Arid Environments*, 41(3):231–45.

Hiernaux, P & Fernández-Rivera, S. 1995. Grazing effects of goat-sheep mixes on the vegetation structure and productivity of old fallows in the Sahel, in *Rangelands in a sustainable biosphere: Proceedings of the Vth International Rangelands Congress*, vol 2, edited by N West. Denver, Colorado: Society for Range Management:230–1.

Hiernaux, P & Fernández-Rivera, S. 1999. Grazing rotation in annual dominated rangelands in the Sahel, in *People and rangelands: Building the future: Proceedings of the VI International Rangelands Congress, Townsville, Australia*, edited by D Eldridge & D Freudenberger:513–4.

Hiernaux, P & Hérault, B. 2003. Dynamics of annual rangelands in the Sahel: Impact of cattle grazing, *Proceedings of the VIIth International Rangelands Congress, Durban, South Africa* edited by N Allsopp, A Palmer, S Milton, K Kirkman, G Kerley, C Hurt & C Brown: 356–8.

Hiernaux, P & Turner, M. 1996. The effect of clipping on growth and nutrient uptake of Sahelian annual rangelands. *Journal of Applied Ecology*, 33:387–99.

Holechek, J, Pieper, R & Herbel, C. 1989. *Range management principles and practices*. Englewood Cliffs, New Jersey: Prentice-Hall.

Huston, M. 1979. A general hypothesis of species diversity. *American Naturalist*, 113:81–101.

Ickowicz, A. 1995. *Approche dynamique du bilan fourrager appliquée à des formations pastorales du Sahel Tchadien*. Thèse Université Paris XII, Créteil, France.

Illius, A & O'Connor, T. 1999. When is grazing a major determinant of rangeland condition and productivity? *People and rangelands: Building the future: Proceedings of the VI International Rangelands Congress, Townsville, Australia,* edited by D Eldridge & D Freudenberger:419–24.

Loehle, G. 1989. Catastrophe theory in ecology: A critical review and an example of the butterfly catastrophe. *Ecological Modelling*, 49:125–52.

Marty, A. 1993. La gestion des terroirs et les éleveurs: un outil d'exlusion ou de négociation. *Tiers Mondes*, 34(134):53–9.

McCoock, L. 1994. Understanding ecological community succession: Models and theories: A review. *Vegetatio*, 110:115–47.

Mentis, M, Grossman, D, Hardy, M, O'Connor, T & O'Reagain, P. 1989. Paradigm shifts in South African range science, management and administration. *South African Journal of Science*, 85:684–87.

Miehe, S. 1998. Inventaire et suivi de la végétation dans le périmètre expérimental à Windou Thiengoly dans le cadre du Projet Sénégalo-Allemand d'autopromotion pastorale dans le Ferlo (PAPF). Saint-Louis, Sengal: GTZ.

Milchunas, D & Lauenroth, W. 1993. Quantitative effects of grazing on vegetation and soils over a global range of environments. *Ecological Monographs*, 63(4):327–66.

Niamir-Fuller, M. (ed). 1999. *Managing mobility in African rangelands. The legitimization of transhumance*. London: Food and Agriculture Organisation/Intermediate Technology.

Nolan, T, Connolly, J, Guillon, L, Sall, C, Dieye, K & Guerin, H. 1989. Mixed animal species grazing under temperate and semi-arid conditions, in *Proceedings of the XVI International Grasslands Congress, Nice, France.*

Oba, G. 1998. Effects of excluding goat herbivory on *Acacia tortilis* woodland around pastoralist settlements in northwest Kenya. *Acta Oecologica*, 19(4):395–404.

O'Connor, T & Pickett, G. 1992. The influence of grazing on seed production and seed banks of some African savanna grasslands. *Journal of Applied Ecology*, 29:247–60.

Odum, E. 1971. *Fundamentals of ecology*. 3rd ed. Philadelphia: Saunders.

Oesterheld, M & McNaughton, S. 1991. Effect of stress and time for recovery on the amount of compensatory growth after grazing. *Oecologia*, 85:305–13.

Poissonnet, J & César, J. 1972. Structure spécifique de la strate herbacée dans la savane à palmier rônier de Lamto. *Annales de l'Université d'Abidjan*, 5(1):577–601.

Rietkerk, M, Ketner, P, Stroosnijder, L & Prins, H. 1996. Sahelian rangeland development; a catastrophe? *Journal of Range Management*, 49(6):512–9.

Rietkerk, M & Van de Koppel, J. 1997. Alternate stable states and threshold effects in semi-arid grazing systems. *Oikos*, 79(1):69–76.

Roberty, G. 1946. *Les associations végétales de la vallée moyenne du Niger*. Zürich: Veröff Geobotanical Institute:1–168. (Veröffentlichungen des Geobotanischen Institutes der ETH, Stiftung Rübel, Zürich; no. 22.)

Schlecht, E, Hiernaux, P & Turner, M. 2001. Mobilité régionale du bétail: nécessité et alternatives? In *Elevage et gestion de parcours au Sahel, implications pour le développement*, edited by E Tielkes, E Schlecht & P Hiernaux. Stuttgart: Verlag E Grauer:291–302.

Scoones, I. 1995. New directions in pastoral development in Africa, in *Living with uncertainty: New directions in pastoral development in Africa*, edited by I Scoones. London: Intermediate Technology:1–36.

Smith, E. 1988. Successional concepts in relation to range condition assessment, in *Vegetation science applications for rangeland analysis and management*, edited by P Tueller. Dordrecht: Kluwer:113–33.

Tausch, R, Wigand, P & Burkhardt, J. 1993. Viewpoint: Plant community thresholds, multiple steady states, and multiple successional pathways: legacy of the Quaternary? *Journal of Range Management*, 46:439–47.

Thébaud, B. 1990. Politiques d'hydraulique pastorale et gestion de l'espace au Sahel. *Cah. Sci. Hum*, 26(1–2):13–31.

Thébaud, B, Grell, H & Miehe, S. 1995. *Recognising the effectiveness of traditional pastoral practices: Lessons from a controlled grazing experiment in northern Sénégal*. London: International Institute for Environment and Development. (IIED paper; no. 55.)

Tilman, D. 1990. Constraints and tradeoffs: towards a predictive theory of competition and succession. *Oikos*, 58:3–15.

Tongway, D & Ludwig, J. 1994. Small-scale resource heterogeneity in semi-arid landscapes. *Pacific Conservation Biology*, 1:201–8.

Trochain, J. 1940. *Contribution à l'étude de la végétation du Sénégal*. Paris: Larose.

Tueller, P. 1973. Secondary succession, disclimax, and range condition standards in desert shrub vegetation, in *Arid shrublands: Proceedings of the 3rd workshop US/Australia Rangelands Panel, Tucson, Arizona*:57–65.

Turner, M. 1998. Long-term effects of daily grazing orbits on nutrient availability in Sahelian West Africa: 1. Gradients in the chemical composition of rangeland soils and vegetation. *Journal of Biogeography*, 25:669–82.

Vedeld, T. 1992. *Local institution-building and resource management in the West African Sahel*. London: Overseas Development Institute. (ODI Pastoral Development Network network paper; no. 33c.)

Walker, B. 1986. A general model of savanna structure and function, in *Determinants of tropical savannas*, edited by B Walker. Lyneham, Australia: Commonwealth Scientific and Industrial Research Organisation:1–12.

Walker, B. 1988. Autoecology, synecology, climate and livestock as agents of rangeland dynamics. *Australian Rangelands Journal*, 10:69–75.

Westoby, M, Walker, B & Noy-Meir I. 1989. Opportunistic management for rangelands not at equilibrium. *Journal of Range Management*, 42:266–74.

Whittaker, R. 1953. A consideration of climax theory: The climax as a population and pattern. *Ecological Monographs*, 23:41–78.

Wiens, J. 1984. On understanding a nonequilibrium world: Myth and reality in community patterns and processes. *Ecological communities: conceptual issues and the evidence*, edited by D Strong, D Simberloff, L Abele & A Thistle. Princeton, New Jersey: Princeton University Press:439–58.

ECOLOGICAL, HISTORICAL AND SOCIAL PERSPECTIVES ON RANGELAND DEGRADATION IN ARID COMMUNAL RANGELANDS IN NAMIBIA

DAVID WARD

INTRODUCTION

Conventional wisdom views heavy grazing by communal pastoralists (subsistence livestock ranching with no private ownership of land) as the major cause of desertification in semi-arid and arid areas of Africa (see for example, Acocks 1953; Jarman & Bosch 1973; Sinclair & Fryxell 1985; Middleton & Thomas 1997). A conventional view of communal pastoralism is that there is little or no management or protection of the resource being used. This lack of resource management may lead to what Hardin (1968) termed the 'tragedy of the commons'. That is, because the grazing resource belongs to nobody, nobody cares for it. This attitude promotes a selfish strategy where individuals consider only their own potential gain even if this should be at the cost of others. Communal pastoralism is frequently parodied as the classic example of the 'tragedy of the commons' – hence my emphasis on it here.

THE CONTRIBUTION OF ECOLOGY TO RANGELAND MANAGEMENT

> *We are not concerned with the very poor. They are unthinkable, and only to be approached by the statistician or the poet.* – EM Forster

Ongoing land reform and the restoration of land to people forcibly removed during the apartheid and colonial periods has made the issue of the sustainable management of communal ranching lands a prominent and often controversial topic in South African agriculture. Indeed, the importance of this issue in agricultural, economic, conservation and social terms can hardly be understated – 12% of South Africa is currently communally owned. This land supports about 25% of South Africa's people and contains about half the livestock population (De Bruyn & Scogings 1998). I believe that ecologists have a lot to offer to the management of communal rangelands. Ecology is an integrative discipline, with population and community ecology as its cornerstones. These cornerstones are the most important tools in rangeland management, but they are insufficient for the '*compleat* ecologist'. Increasingly, use of remote sensing (both from satellite and aerial photographs) is becoming a key skill because of the need to assess biodiversity over vast areas and to track historical changes in ecosystems (for example, Aharoni & Ward 1997). These are the major tools of the conservation ecologist, particularly in North America and Europe. The African ecologist needs these tools and more. The importance and complexity of social issues in land management and conservation in southern Africa means that the incorporation of a social perspective is of great importance. In the case study outlined below, I attempt to show how ecological, historical and social issues can be integrated in African rangeland ecology.

DOES HEAVY STOCKING BY COMMUNAL PASTORALISTS INEVITABLY LEAD TO LAND DEGRADATION ?

> *A great many people think they are thinking when they are merely rearranging their prejudices.* – William James

Communal pastoralism is widely considered to have very negative effects on the environment for the reasons outlined by Hardin (1968). Such claims are far less commonly made for the effects of commercial ranching because it is assumed that single farmers (or corporations) must make rational decisions based on the potential benefits of a particular management strategy, with the awareness that all of the costs of such decisions

are borne by themselves. However, such costs may be difficult for the commercial farmer to predict ahead of time. For example, variation in rainfall will affect grass biomass, which in turn will affect optimal stocking rates. Miscalculations of stocking rates may have serious negative consequences for grass production and biodiversity in the long term. Rainfall prediction is still an imperfect science; consequently, stocking rate calculations may be poorly synchronised with the ability of the environment to support those livestock. Additionally, short-term profits may be an important goal for the commercial farmer, regardless of possible long-term costs. Hence, the assumption that commercial ranches will be more sustainably managed because there is private control is not as convincing as it initially appears. Ascertaining whether the ‘tragedy of the commons’, rather than some other management or environmental process, is responsible for environmental degradation can be done by direct comparison of the effects of commercial and communal ranching on the environment. This seems, at first sight, to be elementary. However, such comparisons are fraught with difficulty. In South Africa and Namibia, social engineering during the colonial and apartheid eras frequently exploited differences in initial environmental quality to the benefit of commercial farmers by forcibly removing communal farmers to areas of lower agricultural potential. Consequently, attempts to compare the effects of communal and commercial ranching are confounded by differences in environmental quality *prior* to the establishment of the respective management strategies. The communal ranching area, Otjimbingwe, in arid central Namibia (mean annual rainfall = 165mm), is an exception in this regard because communal ranchers have lived there for several hundred years and it is completely surrounded by commercial (privately-owned) ranches. This allowed us to examine the relative effects of communal and commercial ranching on biodiversity and soil quality under similar environmental conditions.

There has been a huge increase (about 500–800%) in human population density in the communal ranching area of Otjimbingwe since the 1950s. Heavy grazing is apparent, particularly in frequent dry years. Furthermore, there has been an apparent change in the type of livestock ranched, from cattle (predominantly) to goats and some sheep. This change is associated with a change in the human population. Otjimbingwe was once almost entirely populated by Herero people, who are largely cattle ranchers. However, today the population consists of Herero and Damara people, who are represented in approximately equal proportions. The Damara people ranch mostly with goats. This change indicated that, perhaps, land degradation had occurred to such an extent that the Herero people found it difficult to raise large grazing animals such as cattle, and that only the small browsing/grazing animals such as goats could be sustained. Hence, the change in human populations may reflect declining environmental conditions. If so, this would constitute evidence of land degradation or desertification of the area. Furthermore, management of grazing by the local headmen or community groups does not occur in Otjimbingwe, *contra* observations in other African communal areas (see for example, Ellis & Swift 1988). Thus, Otjimbingwe appeared to have the potential for the ‘tragedy of the commons’ to occur.

Another reason for choosing Otjimbingwe for this study was that it had a long written history owing to the presence of missionaries and a trader, Charles John Andersson, who wrote extensive diaries of conditions there 150 years ago. Such records are invaluable to the ecologist wishing to determine whether there has been a change in environmental quality in the long term.

In spite of the absence of an organized management strategy, and extremely high stocking densities, we found no long-term degradation of vegetation or soil resources (Ward et al. 1998). We found that plant cover, species richness and diversity as well as grass availability after a rainy season is similar to that on surrounding commercial ranches, which have approximately tenfold lower stocking densities. Similarly, we found no significant difference in key soil parameters between Otjimbingwe and the surrounding commercial ranches (which have been in use in this century only) (Ward et al. 1998). The only evidence we have of land degradation is from the decline in grass production at Otjimbingwe watering points used for over 150 years (as listed by Andersson in 1856 & 1864) relative to the new watering points in use for less than ten years. Furthermore, if there had been serious land degradation we would expect to see a decline in stock numbers over time as a consequence of insufficient soil resources to maintain forage resources (for example, Dean & MacDonald 1994). However, there has been no noticeable decline in stock numbers, substantiating our claim that no serious land degradation has occurred. Our findings are consistent with those of Ward and Ngairorue (2000) and Ward et al. (In press) who showed that heavy grazing does not usually cause degradation in Namibia’s arid environments in the short- to medium-term. Heavy grazing may only cause land degradation in arid regions if it is sustained in the long term, say in excess of 80–100 years (see also Wiegand & Milton 1996). Furthermore, there is generally a positive correlation between the effects of grazing and mean annual precipitation in African ecosystems receiving less than 400mm annual rainfall (Milchunas & Lauenroth 1993; Ward et al. 2000a).

The Otjimbingwe case study is not an isolated example: in spite of five to ten fold higher stocking rates on communal ranches, few of the large number of African studies have shown differences in effects on biodiversity, plant species composition and soil quality between

commercial and communal ranches (for example, Archer et al. 1989; Tapson 1993; Scoones 1995; Ward et al. 1998, reviewed by Behnke & Abel 1996). Consequently, the 'tragedy of the commons' concept has been heavily criticized in recent years, by Ellis and Swift (1988), Archer et al. (1989), Shackleton (1993), Werner (1994), and Scoones (1995), among others. The criticisms of the 'tragedy of the commons' paradigm in the context of communal pastoralism in arid rangelands are threefold:

- In spite of communal ownership and lack of fencing of separate grazing areas, many communal ranches have a high degree of management by local communities. In many communal areas, tribal leaders and/or community groups decide on who may graze where and how long they may use the resource.
- The high inherent variability of many arid African pastoral ecosystems is driven by the availability of rainfall. Consequently, the effect of high stocking rates on the subsequent availability of vegetation to livestock is frequently rather small in comparison with the effects of abiotic variables such as local rainfall. Rainfall is seen as having the ability to 'recharge' a system that suffers heavy grazing pressure. Overgrazing, in this view, is therefore usually a short-term problem that is frequently rectified by large rainfalls in some years. Indeed, it is generally agreed that where pastoralists are able to maintain their activities on a large spatial scale by migrating to areas where key high resources can be exploited, allowing previously-used resources time to recover, negative effects of grazing on plant biodiversity do not develop (Sinclair & Fryxell 1985; Ellis & Swift 1988; Behnke & Abel 1996). An additional issue of importance is a logistic one: it is very difficult to distinguish *signal* (effects of herbivory) from *noise* (inherent ecosystem variability) in arid ecosystems (Ward et al. 2000b). However, this means neither that it is impossible to do so nor that effects of herbivory are unimportant.
- Livestock of communal ranchers die off in large numbers during sustained drought. Thereafter, these herds slowly recover, allowing the forage plants to recover lost resources after the drought. Contrastingly, commercial ranchers maintain their herds via supplementary feeding and purchase of stock, thereby sustaining the pressure on the vegetation during the vital post-drought recovery period. Ultimately, these two strategies may have similar negative effects on the environment. Put another way, the consequences of the 'tragedy of the commons' may be no worse than the consequences of shortsightedness or greed on the part of the individual private landowner.

I wish to stress that the people of Otjimbingwe have suffered considerable erosion of their lifestyles over the last 100 years. However, this has relatively little to do with communal pastoralism. For example, Otjimbingwe residents annually produced up to 95 tons of wheat in the ephemeral Swakop river bed in the 1920s. Today, no wheat is produced on account of the rarity of river flow – two dams were built upstream for the purpose of supplying Windhoek and Okahandja with water and to 'manage' river flow. However, only two substantial flows have been recorded in the 20-odd years since the dams were built on account of the high water requirements of the above-mentioned urban areas (Ward et al. 2000c). This type of external limitation on environmental quality and lifestyle of rural people is sadly all too frequent and clouds this contentious issue, making it all the more difficult for the observer to ascertain the causes of environmental degradation in communal areas.

The Otjimbingwe study taught me a number of things that influence my current approach to communal ranching systems:

- Most importantly, while the concept of long-term sustainability is difficult enough to convince rich people to consider, it is an impossible goal for poor people. Poor people cannot afford to wait until the long term because survival is a day-to-day decision.
- Scientists need to develop management strategies to help communal farmers that optimise short-term objectives and that minimize long-term costs to the environment. However, we must recognize that adaptive management only works up to a point. Ranchers cannot alter their herd sizes to perfectly match their environments because they need to maintain minimum breeding herd sizes for the next wet season. Hence, they will inevitably 'overgraze' in dry seasons.
- Stocking rate is less important than the period of recovery after drought. We need to approach this issue more from the perspective of the resilience of plants to herbivory and less from the perspective of herd management.
- It is clear that many communal farmers are constrained in their actions not only by their local environments but also by national policies and actions beyond their control. The 'tragedy of the commons', if it occurs, may pale into insignificance in comparison with these larger-scale constraints. Consequently, ecologists and rangeland managers need to pay attention to both the local and larger-scale constraints in their research and extension programmes. This is often a hard pill to swallow because we are more comfortable at the local scale, where ecological research and rangeland management traditionally place their emphasis.

REFERENCES

Acocks, J. 1953. Veld types of South Africa. *Memoirs of the Botanical Survey of South Africa,* 28:1–192.

Aharoni, B & Ward, D. 1997. A new predictive tool for identifying areas of desertification: A case study from

Namibia. *Desertification Control Bulletin,* 31:12–18.

Archer, F, Hoffman, M & Danckwerts, J. 1989. How economic are the farming units of Leliefontein, Namaqualand? *Journal of the Grassland Society of Southern Africa,* 6:211–15.

Behnke, R & Abel, N. 1996. Revisited: The overstocking controversy in semi-arid Africa. *World Animal Review,* 87:4–27.

Dean, W & MacDonald, I. 1994. Historical changes in stocking rates of domestic livestock as a measure of semi-arid and arid rangeland degradation in the Cape Province, South Africa. *Journal of Arid Environments,* 26:281–98.

De Bruyn, T & Scogings, P. 1998. *Proceedings of symposium on policy-making for the sustainable use of southern African communal rangelands.* Alice: Department of Livestock and Pasture Science, University of Fort Hare.

Ellis, J and Swift, D. 1988. Stability of African pastoral ecosystems: Alternate paradigms and implications for development. *Journal of Range Management,* 41:450–9.

Hardin, G. 1968. The tragedy of the commons. *Science,* 162:1243–8.

Jarman, N & Bosch, O. 1973. The identification and mapping of extensive secondary invasive and degraded ecological types (test site D), in *To assess the value of satellite imagery in resource evaluation on a national scale*, edited by O Malan. Pretoria: Council for Scientific and Industrial Research:77–80

Middleton, N & Thomas, D. 1997. *World atlas of desertification.* (2nd ed.) London: Edward Arnold.

Milchunas, D & Lauenroth, W. 1993. Quantitative effects of grazing on vegetation and soils over a global range of environments. *Ecological Monographs,* 63:327–66.

Scoones, I. 1995. Contrasting savanna dynamics: Implications for livestock populations in Zimbabwe's dryland communal areas, in *Sustainable land management in semi-arid and subhumid regions*, edited by G Campbell. Proceedings of the SCOPE workshop, Dakar, Senegal. Montpellier, France. Centre de coopération internationale en recherche agronomique pour le développement (CIRAD):353–60.

Shackleton, C. 1993. Are the communal grazing lands in need of saving? *Development Southern Africa,*10:65–78.

Sinclair, A & Fryxell, J. 1985. The Sahel of Africa: Ecology of a disaster. *Canadian Journal of Zoology,* 63:987–94.

Tapson, D. 1993. Biological sustainability in pastoral systems: The kwaZulu case, in *Range ecology at disequilibrium: New models of natural variability and pastoral adaptation in African savannas*, edited by R Behnke, I Scoones & C Kerven. London: Overseas Development Institute/International Institute for Environment and Development/Commonwealth Secretariat:118–35.

Ward, D, Ngairorue, B, Kathena, J, Samuels, R & Ofran, Y. 1998. Land degradation is not a necessary outcome of communal pastoralism in arid Namibia. *Journal of Arid Environments*, 40:357–71.

Ward, D & Ngairorue, B. 2000. Are Namibia's grasslands desertifying? *Journal of Range Management,* 53:138–44.

Ward, D, Ngairorue, B, Karamata, J & Kapofi, I. 2000a The effects of communal and commercial pastoralism on vegetation and soils in an arid and a semi-arid region of Namibia in *Vegetation science in retrospect and perspective*, edited by P White, L Mucina, J Leps & E van der Maarel. Uppsala, Sweden: Opulus Press:344–7.

Ward, D, Saltz, D & Olsvig-Whittaker, L. 2000b. Distinguishing signal from noise: Long-term studies of vegetation in Makhtesh Ramon erosion cirque, Negev desert, Israel. *Plant Ecology,* 150:27–36.

Ward, D, Ngairorue, B, Apollus, A & Tjiveze, H. 2000c. Perceptions and realities of land degradation in arid Otjimbingwe, Namibia. *Journal of Arid Environments,* 45:337–56.

Ward, D, Saltz, D & Ngairorue, B. In press. Temporal and spatial variation in rainfall and grass production and its effects on the management strategies of ranchers in arid Namibia. *Journal of Range Management.*

Werner, W. 1994. Land tenure issues in Namibia: Interactions with sustainable resource management in *Proceedings of Namibia's national workshop to combat desertification*, edited by S Wolters. Windhoek, Namibia: Desert Research Foundation of Namibia:27– 41.

Wiegand, T & Milton, S. 1996. Vegetation change in semi-arid communities: Simulating probabilities and time scales. *Vegetatio,* 125:169–83.

THE INFLUENCE OF COLD TEMPERATURES AND SNOWSTORMS ON RANGELANDS AND LIVESTOCK IN NORTHERN ASIA

CAROL KERVEN

> *The terrible snowstorms which sweep over the northern portion of Eurasia in the later part of the winter, and the glazed frost that follows them;... the heavy snowfalls early in October which eventually render a territory as large as France and Germany absolutely impracticable for ruminants and destroy them by the thousand... these were the conditions under which I saw animal life struggling in Northern Asia. They made me realise at an early date the overwhelming importance in Nature of what Darwin described as 'the natural checks to over-multiplication'* (Prince Petr Alekseevich Kropotkin, 1842–1921 in Todes 1989:128–9).

INTRODUCTION[1]

The climate of much of northern Asia is dry with very cold, often snowy winters. Vast areas of land that cannot be used for agriculture due to cold and/or low rainfall have been used for millennia by nomadic pastoral people to graze livestock (Barfield 1993; Khazanov 1984). Comparable nomadic livestock production systems are also found in the tropical and subtropical dry rangelands of sub-Saharan Africa, where cold winters do not occur. A dry climate and livestock mobility are common to both regions, but the cold factor poses particular challenges for Asian pastoral livestock management.

It has been postulated that the climatic variability and dynamic ecosystem functions of Asia's cold winter rangelands may mean that they are subject to similar non-equilibrium forces as warm semi-arid rangelands in Africa and Australia (Fernadez-Gimenez & Allen-Diaz 1999; Miller 1997). The question had been raised earlier '... under what climatic conditions are these non-equilibrial dynamics likely to occur?... What about dry, cold regions where grazing systems are subject to blizzards rather than or in addition to droughts?' (Ellis et al. 1993:39). This paper considers whether a cold-winter climate extends the applicability of non-equilibrium dynamics developed for the warm semi-arid southern hemisphere rangelands to the Asian cold semi-arid rangelands.

There have been few attempts to compare rangeland ecology and pastoral grazing systems between Asian cold rangelands and the warm rangeland regions of the southern hemisphere. This is due in part to past ideological and persistent linguistic barriers. The scientific and ethnographic material has not been easily accessible to the scholars of these respective regions.[2] Moreover, our specialised geographical foci have sometimes prevented us from looking outwards to unfamiliar territories from the regions we know. The comparison is further complicated by major political and economic differences between sub-Saharan Africa and northern Asia. Most north Asian pastoralists have been incorporated into centralised state production systems for more than half a century, while most African pastoralists have been integrated into market economies for at least as long.

The model of non-equilibrium dynamics for semi-arid warm rangelands received considerable attention from donors and researchers concerned with pastoral Africa (for example, De Haan 1994; De Haan et al. 1997; Delgado et al. 1999; Niamir-Fuller 1999; McCarthy et al. 2000). Field tests of the model have been carried out mainly in Africa while its applications to policy are still debated.[3] A central notion of the model is that unpredictable climatic conditions, principally variability of precipitation, result in a de-coupling of livestock populations from their vegetation resource base. Perhaps curiously, there has not been a rush to test this model in other rangeland regions of the world that are also subject to climatic instability.

This paper does not offer any new evidence from north Asian field studies of vegetation and precipitation interactions, as has been tested for example by Fernadez-

Gimenez and Allen-Diaz for the Mongolian case (1999). Rather, it is proposed that Asian cold winters with attendant blizzards create a similar climate stress on livestock populations and therefore on pastoralists as is imputed to severe droughts in the dry rangelands of the southern hemisphere. It is also argued that political and economic interventions have profoundly mitigated the short-term effects of climatic influence on livestock, but may have negative effects on northern Asian rangelands over the long term. Whether the non-equilibrium model derived from warm dry ecosystems is generally applicable to cold dry ecosystems of northern Asia remains to be more widely and thoroughly investigated.

Case material is drawn from the cold rangelands in central Asia under the former Soviet Union, Soviet-influenced Mongolia, northern China and the Tibetan plateau cold rangelands. Central Asia here refers to the five former Soviet Republics of Kazakstan, Turkmenistan, Uzbekistan, Kyrgyzstan and Tajikistan. Inner Asia refers to Mongolia, Inner Mongolia and parts of Xinjiang Province in China, as well as portions of the Russian Federation in southern Siberia. The Tibetan plateau is considered separately. This somewhat disparate group of contiguous regions and cultures can be collectively termed as the northern Asian rangelands characterised by extended cold winters with snow.

The areas and characteristics of northern Asian rangelands are outlined next, followed by a discussion of how cold temperatures affect rangeland and livestock management in these regions. We then consider the question of whether the Asian cold semi-arid rangelands are subject to similar non-equilibrium dynamics as has been posited for warm semi-arid rangelands. Some policy and management implications of the non-equilibrium model for northern Asia are given in conclusion.

THE EXTENT AND CLIMATE OF COLD-WINTER NORTH ASIAN RANGELANDS

The northern Asian rangelands span nearly 7 000km. from Kalmykia in the west to eastern Siberia (see map, Figure 1). Together they comprise 25% of the world's total rangelands (Table 1) and over 6% of the total world land area. The major proportion of this region has very cold winters, although the total area subject to cold winters is difficult to estimate as the degree of cold depends on latitude and altitude. The definition of cold-limited northern rangelands is areas where winter temperatures stop plant growth for more than a few days each year (Babaev & Orlovsky 1985).

In Central Asia the rangelands north of latitude 43° are broadly classified as northern desert and steppe regions (Babaev & Orlovsky 1985). Together with the mountain pastures in the Tien Shan, Altai and Pamirs, some three quarters of the rangeland can be considered to be within the cold-winter zone (Gilmanov 1996). The northern rangelands of Kazakstan have an average winter temperature of –2°C to –12°C (Gilmanov 1996). Winter snow depth is around 30mm in the semi-desert zone and higher in the more northerly steppes (Robinson 2000). The rangelands of southern Kazakstan, much of Uzbekistan and Turkmenistan do not normally experience winters cold enough to limit plant growth (Babaev & Orlovsky 1985).The Pamir rangelands in Tajikistan have a mean January temperature of –18°C, absolute minimum temperatures of –45°C, annual precipitation of only 73mm, frequent heavy snowfalls and only five relatively frost-free months per year (Government of Tajikistan 1999).

In China, the northern rangelands which are subject to extreme winter temperatures and short growing seasons begin at around latitude 40° north and comprise more than half of all China's rangelands (CSCPRC 1992). The high Tibetan plateau alone constitutes 42% of China's rangelands, with 120 million ha (Wu Ning 1997). Although lying south of latitude 40° north, the cold winters can be more extreme in these highlands. The mean winter temperature is –10°C and plant growing seasons only last between three and six months (Wu Ning 1997).

Mongolia's rangelands all fall into the cold-winter category, lying between latitudes 41° and 52°. Winter temperatures are lowest in the northern mountains with means of –30°C, –25°C in the central steppe and –20°C in the southern Gobi region. Maximum snowfall likewise follows a gradient from 20mm in the north to 5mm. depth in the south (Purev 1990).

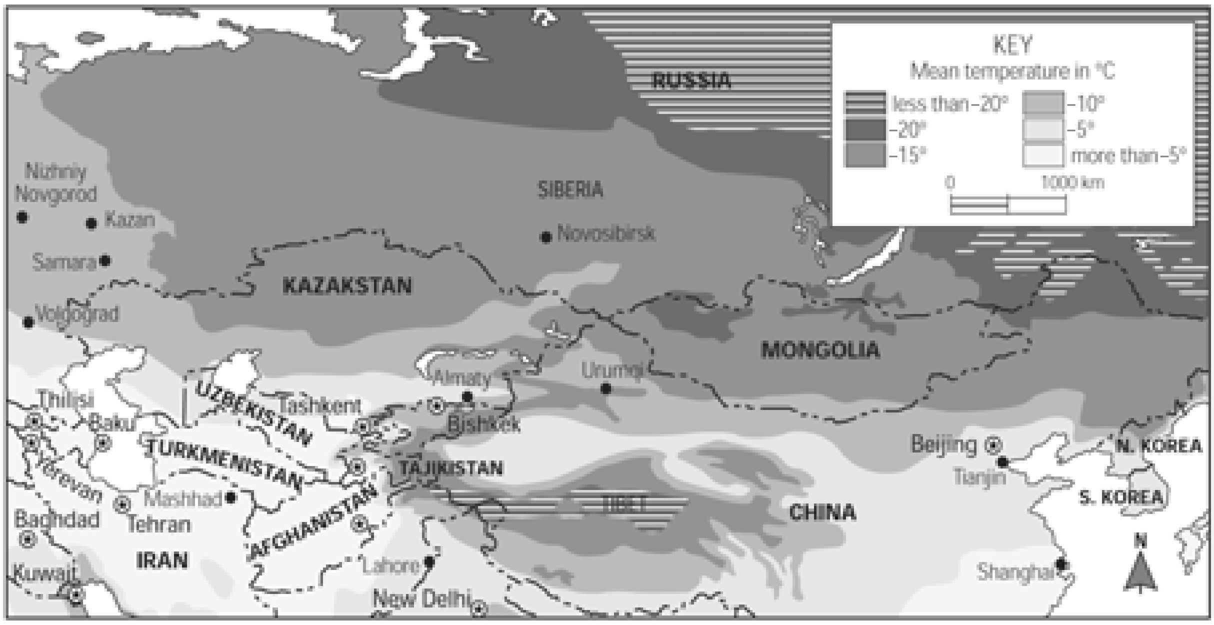
Table 1: Area of northern Asian rangelands (permanent pasture) (Source: FAOStat 1998)

COUNTRY AND REGION	MILLION HECTARE
Russian Federation (including Arctic Siberia)[4]	89.3
Kazakstan	182.3
Turkmenistan	30.7
Uzbekistan	22.8
Kyrgyzstan and Tajikistan	12.7
Mongolia	117
China (including Tibet Autonomous Region)	400
Total northern Asia (= 25% world total rangelands)	854.8
World total permanent pasture	3 426

EFFECTS OF COLD WINTERS ON RANGELAND PRODUCTION AND ON LIVESTOCK

The severity of cold varies with the latitude, altitude, aspect and wind speed. These factors, as well as day length, have a marked effect on range vegetation

Figure 1: Map of countries of north Asia with mean January temperature



composition and productivity, depending on the location and season. Cold winters thus have regular annual impacts on rangeland vegetation and thus on livestock dependent on rangelands. These impacts on range vegetation are outlined next. But cold winters can also have irregular and catastrophic effects on livestock, as is discussed in the following section.

In cold-season environments, winter acts on pasture much like the dry season does in warm tropical systems. There is no growth, so even when the energy content of plant biomass is not lower in the cold (or dry) season, there are less energy-rendering nutrients and thus the quality of the plant biomass is lowered (Coppock et al. 1986). Nevertheless, herbivores manage to survive on this poor quality range so long as the winter does not last too long or forage becomes unobtainable. Animals lose weight and body condition over the cold season. If forage quality is sufficiently impaired, weaker animals will die and females will bear underweight offspring. Thus, reduction of pasture quality during winter inhibits livestock production, much as the dry season in warm tropical rangelands.

Spatial variability of pasture yield and nutritive values is typical of the northern Asian rangelands as in the warm semi-arid rangelands, and likewise underlies the systems of pastoral migration. In both warm and cold rangelands, spatial variability is frequently due to an inverse correlation between plant quality and the quantity of production along a geographical gradient determined by climatic factors (Breman & De Wit 1983). In the northern rangelands, the inverse correlation between plant quality and quantity is due not only to precipitation gradients but the additional factor of temperature, as lower temperatures retard plant maturation in the growing season, increasing plant digestibility for animals (Langvatn et al. 1996). Therefore decreasing temperature and water supply results in better quality but lower pasture quantity.

Northern Asian pastoralists have responded to this seasonal spatial variability by moving livestock across latitudes on north-south meridional routes and across altitudes up and down slopes. As temperatures decrease with altitude in the high frigid rangelands of the Tibetan Plateau, plant yield declines but quality increases (Wu Ning 1997).

In Mongolia's drier southern Gobi rangelands, the xerophytic pasture species contain less moisture and are richer in protein, vitamins and minerals compared to the more succulent species in the wetter and mountainous north (Tserendulam 1990). In Mongolia, pasture yields decrease in quantity from north to south, although pasture quality and soil fertility improves (Purev 1990).

Similar relationships between precipitation, altitude, pasture yield and protein values across locations are found in Kazakstan. The desert species have a higher protein value but lower yield compared to steppe and mountain pastures (Alimaev 2004). Animals migrating to the more arid desert pastures in winter and early spring benefit from the more nutritious vegetation, while higher-yielding mountain pastures would allow animals to fatten over the summer. In the steppe and semi-deserts of Kazakstan, both the quantity and quality of forage for livestock is lowered over winter, where the standing biomass of winter forage is about half of those in summer while digestible protein in winter drops to a quarter of the maximum annual level which occurs in spring (Alimaev 2004).

Dormancy is an important plant adaptation to cold winters. Growth ceases in annuals and many perennials and nutrients are conserved below ground. The winter period of plant senescence and dormancy has been described, for example, for the frigid high altitude zones of the Tibetan plateau (Wu Ning 1997).

In Mongolia, where livestock do not receive much supplementary feed, animals routinely lose up to 30% of their body weight over winter (Purev 1990). The seasonal cycle of low and high pasture productivity is summed up in a Kazak proverb: 'Sheep are fat in the summer, strong in the autumn, weak in the winter and dead by spring' (Barfield 1993:142).

Another usual effect of winters in the higher latitudes or altitudes is that some pastures are inaccessible to animals during winter, due to the depth of the snow or to extreme cold temperatures. The alpine meadow pastures of northern Asia often have more fertile soils, which together with their higher precipitation, results in the production of the best quality and quantity of forage in a pastoral region. These alpine pastures have therefore been the summer destination of migratory pastoralists using seasonal transhumance in response to cold. This pattern is found in upland areas of northern Mongolia and Inner Mongolia (Sneath 1999), the Tibetan plateau (Wu Ning 1997), in the Tien Shen and Altai mountains of western China and eastern Kazakstan (Sneath 1999; Benson & Svanberg 1998), Kyrgyzstan (Mearns 1994; Van Veen 1995), and the Pamirs in Tajikistan (citations in Khazanov 1984).

In steppe regions, animals can graze the colder latitudes in more northern areas during summer where sufficient rainfall results in a flush of vegetation. But grazing animals must vacate the northern steppes and shift southwards to warmer areas or to sheltered refuges for winter to avoid the extreme cold and the risks of heavy snowfalls. This north-south migratory system over long distances was practised by Kazak pastoralists moving up to 1000km on the steppe plains prior to Russian settlement (Federovitch 1973; Robinson 2000), and is still followed in Mongolia (Sneath 1999).

THE ADAPTATION OF DOMESTIC LIVESTOCK TO THE COLD

The indigenous livestock of northern Asia have evolved and been selected through domestication to cope with cold winters. Unlike the warm rangelands of the southern hemisphere, cattle are less numerous than other species, as they are less able to forage in winter when snow covers the ground (Matley 1989). Horses are able to kick the snow aside and forage up to 50cm deep in snow, and were the dominant domestic species of the northern latitudes in Mongolia and Kazakstan. At very high altitudes, yaks replace horses as the best-adapted species to foraging under snow and to thrive under cold conditions, as in the Pamirs and Tibetan plateau (Khazanov 1994).

The indigenous sheep breeds (Ryder 1993; Dmitriev & Ernst 1989) of the steppes and lower altitudes are particularly well-adapted to the harsh winters, with their capacity to store fat in the rump and tail for the winter period of poor nutrition, together with their ability to forage under snow and browse shrubs in winter. The Bactrian camel, with its ability to store fat in its humps for use during periods of scarcity, can also be kept in the colder regions of the Asian steppes (Matley 1989). The reindeer herded in Siberia is physiologically adapted to temperatures as low as –30˚C, surviving by reduction of energy requirements and by efficient use of poor quality forage by heavily modified guts (Krupnik 1993 & Tyler pers. com.).

Livestock of northern Asia such as the yak, Bactrian camel and the cashmere goat have also developed specialised hair and wool characteristics which, by insulation from downy undercoats, enable them to survive cold temperatures.

Through a series of physiological and ecological adaptations, most of the indigenous livestock of northern Asia can survive most winters. Human interventions have greatly increased the survivability of livestock over winter through provision of shelter and fodder. Without these interventions, livestock remain vulnerable to sudden severe climate events, as is discussed below.

COLD-RELATED CLIMATIC EVENTS AND UNSTABLE LIVESTOCK POPULATIONS

A critical cold-related but episodic stress occurs throughout the northern latitudes and higher altitudes of Asia. Winter climatic events – sudden heavy snowfalls and thaws followed by freezing – can have immediate and devastating effects upon animals.

In the deserts and semi-deserts of Kazakstan the combination of climatic conditions known as *dzhut*[5] has been recorded approximately once every ten years (Bekenov et al. 1998:49–50; Khazanov 1984; Olcott 1995). *Dhzut* means that the snow is either deeper than 30cm, denser than 0.25–0.30g/cm^2 or covered by a layer of ice, usually in combination with low temperatures and strong winds. Less serious cases of *dzhut* are even more common, occurring three or four times a decade.

Heavy losses of Kazak livestock due to *dzhut* were common prior to the Soviet reorganization of pastoralism (Sludskii 1963, cited in Robinson 2000). A Kazak proverb sums up the vulnerability of livestock to these climatic events: 'Livestock belongs to any snowstorm or powerful enemy' (cited in Khazanov 1984:74). In the 19th century in some years half to three quarters of livestock were estimated to have perished in

regions experiencing *dzhut* (Khazanov 1984; Olcott 1981). The *dzhut* can cause mass mortality of up to 50% among the saiga, a migratory antelope of the same Kazak plains used by livestock, which cannot forage under a deep snow or ice layer (Bekenov et al. 1998). Often *dzhut* are preceded by spring or summer droughts, so animals are already in poor condition at the start of winter. Major *dzhut* that were particularly devastating for saiga were recorded thirteen times between 1826 and 1951. From 1940 to 1970 *dzhut* were the key limiting factors to saiga populations, although since the 1980s the main factors have been disease, poaching and the activities of agriculture and industry – see Figure 2 (Bekenov et al. 1998).

Kazak livestock in Xinjiang province of western China are also subject to *jut* which were especially damaging in late winter or early spring and could decimate herds (Benson & Svanberg 1998:51). Large herds were more affected as owners of smaller herds could store sufficient winter fodder to sustain their animals for longer. The loss of livestock was such that in pre-revolutionary times, the price of meat would double following a *dzhut*.

Severe snow storms likewise cause large-scale livestock mortality on the Tibetan plateau (Richard & Miller 1998; Miller 2000). A typical saw-tooth pattern of a population is shown in Figure 3. Herd populations decline due to freak snow falls disasters, with losses of up to 28% occurring over several years of bad weather, then recovering again (Goldstein n.d., Wu Ning 1997). In the winter of 1997/98 a few areas lost 70% of their livestock, with some three million animals dying (Miller 2000).

Figure 2: Saiga antelope population and winter storms, 1965–96 (Source: Bekenov et al. 1998)

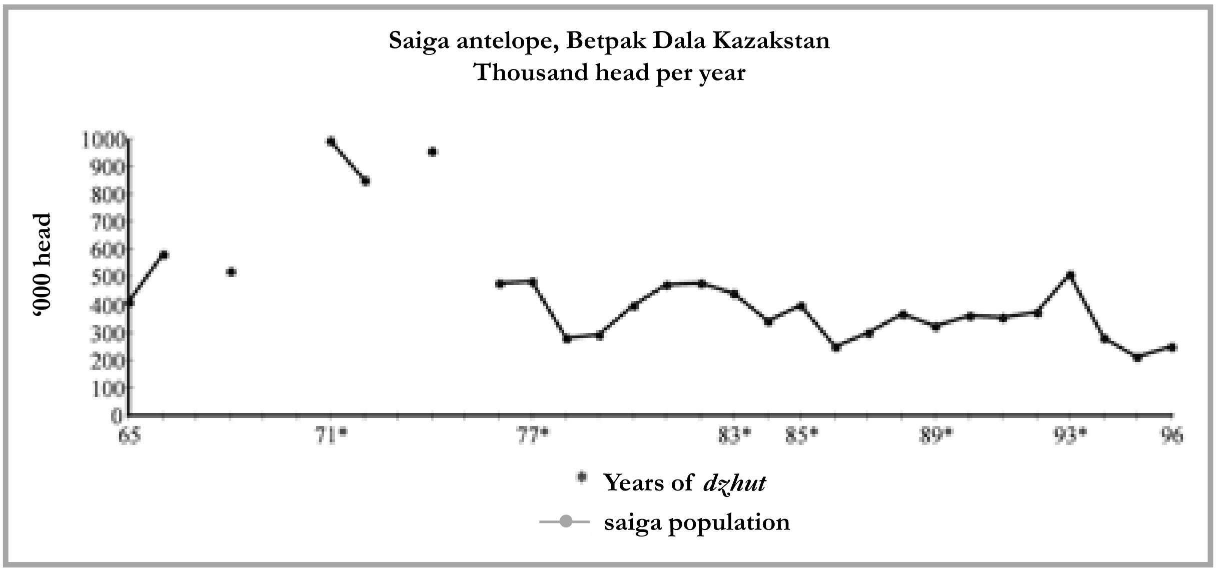
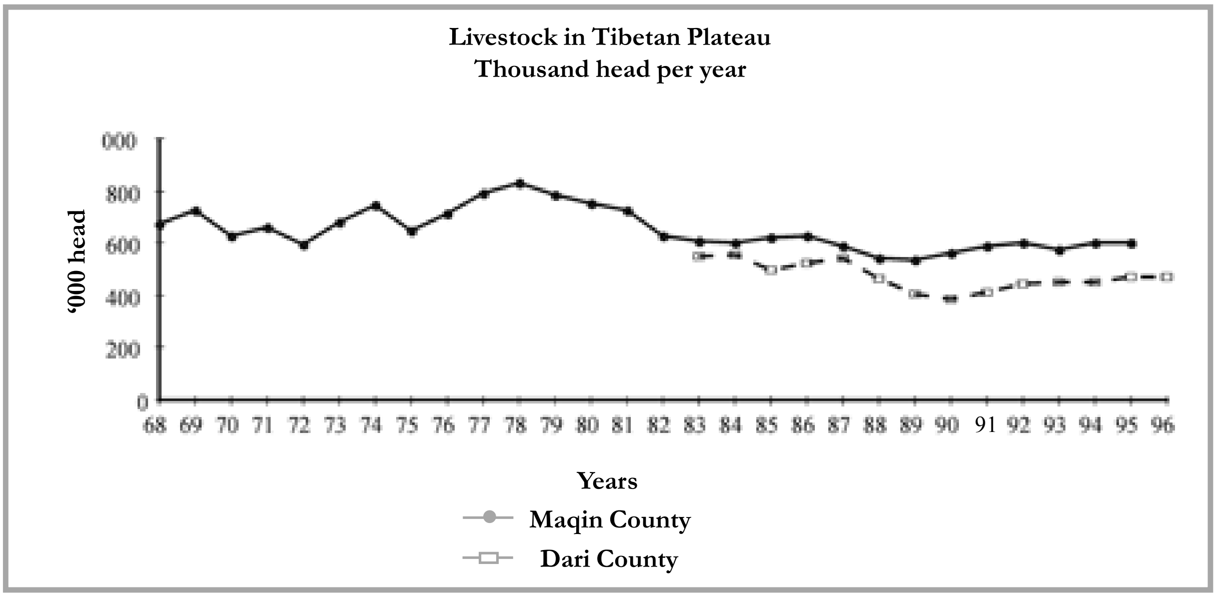


Figure 3: Livestock in Tibetan Plateau, 1968–1996 (Source: Goldstein n.d.)



Mongolia is equally prone to devastating *dzud* (Batjargal et al. 2002). Since the end of Union of Soviet Socialist Republics (USSR) support, there have been four consecutive years in which combinations of drought (black *dzud*), deep snow (white *dzud*) and re-freezing of melted snow (iron *dzud*) have caused millions of livestock to die, lacking state-supported emergency measures, and thousands of pastoral families to lose their livelihoods.

In the tundra and Arctic regions of Siberia, regular collapses of the reindeer populations are linked to patterns of 'ecological disturbances' among which the sudden winter thaw followed by freezing is by far the greatest cause of mass mortality (Krupnik 1993:140;155). In Chukotka, for example, such crashes were recorded every ten to fifteen years from 1880 to 1920. As a result of these huge losses, reindeer pastoralists faced famine.

Massive die-offs of animals due to winter storms could be partly avoided in the past by annual long-distance migrations to warmer latitudes or more sheltered areas, as recorded for Kazakstan (Olcott 1995). In exceptionally snowy winters, movements were extended further southwards to warmer areas beyond the normal winter range, as is illustrated in the case of the saiga antelope in Kazakstan (Bekenov et al. 1998). Kazak pastoralists were also able to extend their movements in the pre-Tsarist period (Olcott 1981; 1995). Similar shifts in migration routes due to climate disasters were reported for Siberian pastoralists prior to Soviet collectivisation (Krupnik 1993).

ARE ASIA'S COLD RANGELANDS A NON-EQUILIBRIUM SYSTEM?

Semi-arid rangelands in Africa are characterised by low and erratic rainfall that has several effects on herbivore populations. Firstly, rangeland net primary production fluctuates widely, with higher productivity in 'good' years, decreasing sharply in 'bad' years. These conditions pertain in rangelands with a coefficient of variation of annual rainfall exceeding 33% – which includes about half the land area of Africa (Ellis 1994a).

Secondly, the non-equilibrium model posits that intermittent but sharp reductions of pasture resources due to drought causes animal mortality to an extent that animal numbers can rarely increase to the point where they would begin to consume more vegetation over their entire range than can be produced over time. According to this model, herbivores in non-equilibrium systems cannot threaten their overall feed supply (though they can and do threaten localised patches of more useful forage or forage around key resources), since ecological carrying capacity is never reached (Caughley cited in Behnke & Scoones 1993; Ellis & Swift 1988; Scoones 1994).

The implication of these processes is that changes in animal population sizes are largely density independent (Ellis & Swift 1988). Population change is heavily influenced by factors other than inexorable internal population growth and consumption of feed resources, leading to insufficient feed per capita and ending in animal die-offs when climatic conditions deteriorate. In the semi-arid tropical and subtropical regions, the main other factor that affects population dynamics is thought to be rainfall. In these areas, herbivore populations rise after a series of good rainfall years, but are then reduced as a result of low rainfall or drought. The balance between animal numbers and feed resources is never stable and therefore these systems are not at equilibrium (Behnke et al. 1993; Scoones 1994; Niamir-Fuller 1999).

Do the rangelands of northern Asia conform to this dynamic found in Africa? On the basis of precipitation alone much of inner Asia could be considered as non-equilibrium according to the warm rangelands model (Sneath 1999:270–2). The southern portion of these rangelands from approximately latitude N35° to N46° has coefficients of variation of annual rainfall exceeding 33%. In these locations, Sneath has argued it is likely that precipitation levels control vegetation output and it is therefore through drought events that livestock experience shortage of feed and populations are affected.

The inter-annual variations in precipitation lead to pronounced and statistically related temporal fluctuations of pasture productivity in the cold-winter rangelands of northern Asia. These fluctuations increase in the drier areas as precipitation decreases. The long-term inter-annual ratio between the maximum and minimum plant mass varies from 3:1 in the wetter regions where meadows and steppes are found, to 10:1 in the more arid zones of southern central Asia (Gilmanov 1995). Even higher ratios have been noted in the semi-deserts of the former Soviet Union, of up to 60:1 for annual grasses (Fedorovich 1973). Winter precipitation as snow also varies from year to year, and as Miller (1997) points out, exceptionally heavy snowfalls can mean greater plant growth in spring time due to increased water infiltration into the soil. This beneficial effect of an unusual precipitation event is in contrast to the disequilibrium effects of droughts in warm rangelands, where plant growth is negatively affected.

Additionally, it has been posited here that in parts of northern Asia climate is a driving variable that has a direct and immediate control on livestock populations. Winter storm events affect animal numbers but not necessarily vegetation output, which may be driven by other climate variables of precipitation and temperature.

Large parts of the northern Asian rangelands experience both variable precipitation and cold events of blizzards and ice freezing. Overlapping and to the north and south of these rainfall-limited rangelands, another crucially variable factor is the degree of cold and snow depth in

winter, in northern Mongolia, Central Asia, Siberia and the Tibetan Plateau. In the most extreme climate of Russian Siberia, it has been noted that 'Arctic ecosystems display periodicity with exceptional clarity' (Krupnik 1993:143).

There are likely to be latitudinal and altitudinal bands in which inter-annual variation in both precipitation and temperature play a role in causing disastrous climatic events for herbivores. This is the case in the southern deserts of Turkmenistan and Uzbekistan, where occasional freak snowstorms can wreak havoc on livestock, while an unusually cold winter results in reduced yield of desert plants in spring (Babaev 1999:11). Shepherds in Turkmenistan's Karakum desert find that below-average spring rainfall is the main regular hazard facing livestock, while sudden snowstorms occur much less frequently (Kerven 1999). These shepherds point out, however, that snowstorms can cause mass animal mortality, compared to low spring rainfall, which just reduces lamb survival and weight gain rates in young animals.

Snow and freezing events cause an interaction between food unavailability and cold, if forage is inaccessible. Since blizzards often immobilize herds, the animals quickly weaken and die from hypothermia (Fernadez-Gimenez 1999). The impact of winter storms on livestock is different from that of drought. In drought animals starve, but in *dzhuts* forage becomes inaccessible due to deep snow cover and/or an icy impenetrable layer. Unable to continue grazing and thereby generate heat through rumen fermentation, digestion and metabolism, the animal's lower critical temperature is exceeded as heat loss is greater than heat gain – 'in simple terms, the more an animal eats the more tolerant it will be to cold' (Webster 1983:644). Livestock freeze to death due to heat loss rather than dying of starvation. The impact of *dzhuts* is immediate and usually cannot be escaped, but droughts take time to build up, giving herders time to move their animals to more abundant pastures, in some cases.

Mass mortality from sudden severe cold-related disasters in northern Asia suggests that livestock populations are heavily influenced by density-independent factors. As noted for one area of Mongolia, the major declines in livestock between 1954 and 1994 can be:

> *accounted for by climatic disasters (termed dzuud) which act as density independent limits on livestock populations. Annual forage bottlenecks in winter and spring, when herders rely entirely on standing dry forage (and in the mountain, cut wild hay) are a more regular and predictable constraint on herd productivity.* (Fernadez-Gimenez 1999:7)

The question whether cold-winter northern rangelands are subject to non-equilibrium dynamics has been explored for the case of the Arctic and sub-Arctic reindeer/caribou grazing areas (Behnke 2000). Evidence is found that particular combinations of inter-seasonal and inter-annual variations in temperature and precipitation have complex but definite impacts on reindeer populations. However, in these high latitudes it is difficult to detect a single dominant impact of climate on vegetation and thus on livestock nutrition comparable to rainfall variability in African and Australian tropical rangelands.

Climatic extremes may have similar effects in otherwise dissimilar geographical regions. In warm low-rainfall regimes, non-equilibrium is more acute as rainfall becomes lower and more erratic, and the impacts on vegetation and animal life more dramatic. Such processes also prevail in northern Asia;

> *As continentality and climatic extremeness intensify, so do the amplitudes of 'disastrous' biological pulsations and the severity of subsistence catastrophes.* (Krupnik 1993:127)

Overall, it seems likely that if the cold-winter rangelands of northern Asia had not been perturbed by the artificial provision of supplementary feed in the latter half of the 20th century, growth of the livestock population would have been continually checked by climatic factors. But the relationship between animal populations and grazing resources in the northern Asian rangelands has been fundamentally altered due to large-scale additions of supplementary feed.

PREVENTION OF WINTER LIVESTOCK MORTALITY IN THE SOVIET STATE LIVESTOCK FARMS

During the Soviet era in the northern parts of Central Asia, use of supplementary fodder became the norm during the winter season of low plant productivity and during periods of exceptional cold and snow cover. Whereas previously livestock populations had naturally been checked through mortality factors induced by cold winters, the Soviet challenge was to meet increasingly higher planned state targets for meat and wool from the rangelands (Channon & Channon 1990; Gilmanov 1995). Provision of supplementary fodder for winter was viewed as the main requirement for increasing animal numbers (Central Asian Review 1954).

The shortage of winter forage caused by rising livestock numbers was addressed by increasing cultivated fodder crops mainly on irrigated land. In Kazakstan winter feed production increased from a couple of million tonnes per year of hay in the mid 1950s to over 30 million tonnes by the mid 1980s, mostly from cultivated forage crops (Alimaev 2004). Furthermore, strenuous measures were taken to boost rangeland vegetation production, and thus livestock output, through phyto-amelioration (Nechaeva 1985). Plans were laid, typically on a heroic scale, to improve over 10 million ha of natural rangelands in central Asia through re-seeding and other measures (Babaev et al. 1991). Across the Soviet Union from 1965 to 1985, the proportion of conserved feed in livestock diets changed from 40 to 55% (Gilmanov 1996).

The effect of adding conserved feed reserves to the pasture resources was to buffer the fluctuations in animal numbers. This can be seen in the smooth upward curve in the livestock population of Kazakstan from the 1940s, reaching a plateau in the late 1970s (Figure 4). If this curve is compared to the jagged pattern of the saiga wild herbivore population in Kazakstan over a comparable period from 1965–96 (Figure 2) it seems clear that the impact of climate on livestock has partly been averted through the use of additional winter feed. However, the impact of veterinary interventions, winter shelters and breeding programmes on increasing livestock numbers should not be ignored, nor should the impact of over-hunting on lowering saiga antelope populations (Milner-Gulland 1994).

POLICY AND MANAGEMENT IMPLICATIONS OF THE NON-EQUILIBRIUM MODEL IN NORTHERN ASIA

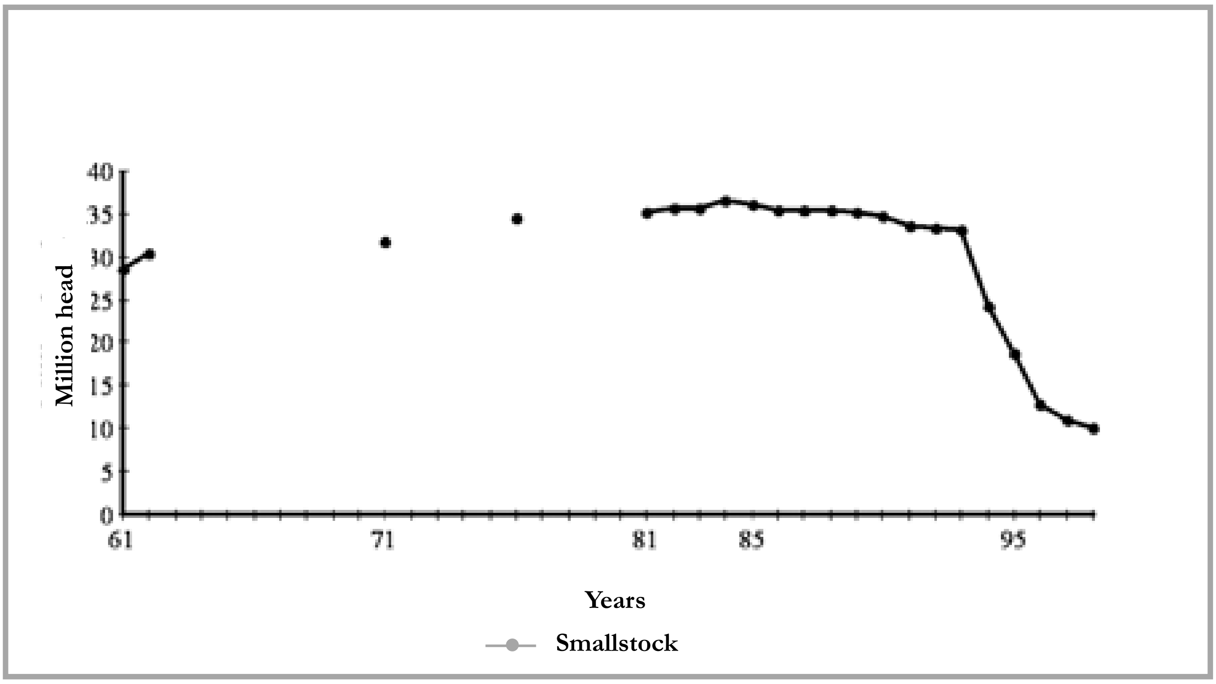
The non-equilibrium model alters not only how we understand rangeland dynamics but also carries implications about how non-equilibrium rangelands should be governed and used (Ellis 1994b; De Haan 1994; De Haan et al. 1997; Niamir-Fuller 1999). Policies towards rangeland management in northern Asia that rest (even unwittingly) on assumptions of climate stability may ignore key management requirements of a non-equilibrium system (Humphrey & Sneath 1999; Miller 1997).

Three management issues arise from the cold winter environments of northern Asia: unstable livestock populations, procurement of winter feed and rangeland property rights.

Non-equilibrium systems raise vexing questions of how to determine the number of animals that can be carried at an adequate level of production on a given area (see Behnke & Abel 1996 for a review). In an ecosystem containing seasonally variable pastures grazed by a population of herbivores, carrying capacity is a function of the most limited seasonal feed resource in the annual cycle. Thus, even if large amounts of feed are routinely available in some seasons – in warm rangelands during the rainy season while in cold-winter rangelands over the summer – livestock numbers are pushed back regularly in the season of vegetation scarcity – the dry or cold season, respectively. Add to this pattern the infrequent but exceptional climatic events that drastically reduce animal populations and the question of how many animals can be kept in a given ecosystem becomes difficult to settle.

Pastoralists in areas subject to periodic heavy mortality among livestock will seek to maximise the numbers of their personal animals, as insurance in the eventuality of losing many animals and being reduced to a below-subsistence level. Due to climate catastrophes in the

Figure 4: Livestock in Kazakstan 1961–98 (Source: Behnke 2004)



cold-winter north Asian rangelands, it has been noted that pastoralists strive for increased production (Khazanov 1984; Krupnik 1993). For this reason, policies that seek to impose maximum stocking rates are imposing considerable risks on pastoralists, if no interventions can be provided by the state in the event of large-scale livestock mortality. Such policies are in fact proving very difficult to implement in China (Banks 1999; Longworth & Williams 1993; Rozelle et al. 1997; Williams 1996). More effective and economically attractive are policies encouraging profitable and accessible marketing channels that allow pastoralists voluntarily to destock when climatic factors threaten (Wu Ning & Richard 1999).

Maximizing livestock numbers is widely perceived as leading to overstocking and rangeland degradation in pastoral Asia, as in Africa. From this perception comes the insistence by the Chinese government that stocking rates by individual herders need to be controlled (Neupert 1999; CSRPRC 1992; Banskota et al.1999). However, studies in northern Asia now suggest that high stocking rates per se are not likely to be the main cause of degradation (Chuluun & Ojima 1999; Goldstein n.d.; Humphrey & Sneath 1999; Williams 1996). Instead, reduction of mobility has been clearly linked to greater levels of rangeland degradation, whether due to sedentarisation of herders, increasing dependence on winter feed, restriction of access to seasonal pastures due to agricultural encroachment, or combinations of these.

It has been argued for tropical semi-arid areas in Africa that if herbivore populations are constantly being checked by droughts, then the risk of overgrazing through unrestricted population growth is limited (Abel & Blaikie 1989; Behnke et al. 1993). This line of argument, however, presupposes that no additional feed resources are brought into the grazing system, as is typical in sub-Saharan Africa. If supplementary feed is provided, then animals can ride out the bad seasons or years without high mortality or reduced offtake, and increasing populations begin to threaten the grazing resource. Such a scenario has been described for north African and Middle Eastern grazing systems where the use of subsidized supplementary fodder has been widespread (Seligman & Perevolotsky 1994).

Providing additional feed to animals when their need is greatest has been one solution to the problem of balancing livestock feed requirements with seasonally-uneven forage supplies. Some pastoral groups in the past achieved this by conserving natural forage as hay at the end of the summer season, for use in the winter season of feed shortage. In modern times, industrial countries, including the former Soviet Union, intensified the practice of supplementary feeding in winter. On the cold rangelands of North America, animals on winter pastures have been supplemented since the 19th century, following the calamitous losses during blizzards when Europeans first introduced cattle into the northern plains (Young & Sparks 1985; Dobie 1980). Under comparable climate conditions in Kazakstan, the Soviet state farms came to depend on winter feed supplements to maintain and increase livestock numbers.

Seasonal movement provides another way of adjusting feed supplies which vary predictably by season and location and unpredictably from year to year due to climatic events. Flexibility to move large distances and in different directions is vital, when rains fail or winters are particularly harsh. Movement in the event of drought is illustrated by the case of Mongolia. The largest pre-revolutionary administrative districts that corresponded to annual grazing areas were those where rainfall variability is highest, in the Gobi region of the south (Sneath 1999). Pre-revolutionary districts had to be large enough to incorporate the maximum distances travelled by pastoralists within each district. In modern times, Mongolian pastoralists in the Gobi desert still routinely move livestock hundreds of kilometres during droughts (Ellis & Chuluun 1993; Sneath 1999; Fernandez-Gimenez 1999).

Non-equilibrium rangelands require appropriate property rights (for recent discussions on Africa, see McCarthy 2000; Niamir-Fuller 1999). Flexible boundaries and communal control are necessary in variable environments where only very large areas of privatised land can encompass sufficient seasonal feed resources to sustain enough livestock to support a family. If only small pieces of rangeland are privatised, the burden of feeding private animals either falls on to the remaining communal lands or ranches become unsustainable. The sizes of fenced ranches in the Americas, Australia and of European colonists in Africa are proof that in non-equilibrium climates, private ownership of rangeland needs to be on a large scale and to exclude other previous users.

The Chinese government has been encouraging individuals and groups to privatise parcels of rangelands. These new forms of rangeland property rights are a response by the government to the successful efforts of pastoralists to increase their livestock numbers following policies to privatise livestock ownership in the late 1970s. The Chinese government and some scientists have attributed the root cause of these changes to a tension between private ownership of livestock and the open access to grazing land following decollectivisation of land in the 1980s. Referred to as the problem of ‘eating from the big rice pot’, this is the familiar logic of ‘the tragedy of the commons’, a process which is perceived as inexorably leading to overstocking, degradation and finally desertification of the rangelands (Banks 1999; CSCPRC 1992; Ho 1996; Wu Ning & Richard 1999).

The Chinese policy rationale was that pastoralists would care more about conserving rangeland if they had individual control over their own land and that rangeland enclosure would 'force independent households to confront the contradictions between forage demand and forage availability among their separate herds' (Williams 1996:308). With encouragement and subsidies from the state, pastoralists are producing supplementary feeds using privatised land on which fodder crops can be grown (Sneath 1999; Neupert 1999; Wu Ning & Richard 1999). The effect of greater access to fodder has allowed more animals to be maintained through the winter, putting increased grazing pressure on spring and summer pastures as livestock populations rise (Reynolds et al. 1999; Williams 1996; Wu Ning & Richard 1999). The key limiting factor of winter forage supply has been overcome with intensification, as previously occurred in Soviet-managed state livestock farms of Central Asia.

The last decade in ex-Soviet central Asia is a complete contrast to the rise of livestock numbers and associated levels of supplementary feed in China's pastoral regions. Policies and market opportunities have prompted China's pastoralists to overcome livestock feed limitations in a cold-winter region. In a buoyant market for livestock and their products, pastoralists can afford to increase stock numbers by giving additional feed to their livestock over the critical winter period of low pasture availability. The effect on livestock populations is shown in Figure 5.

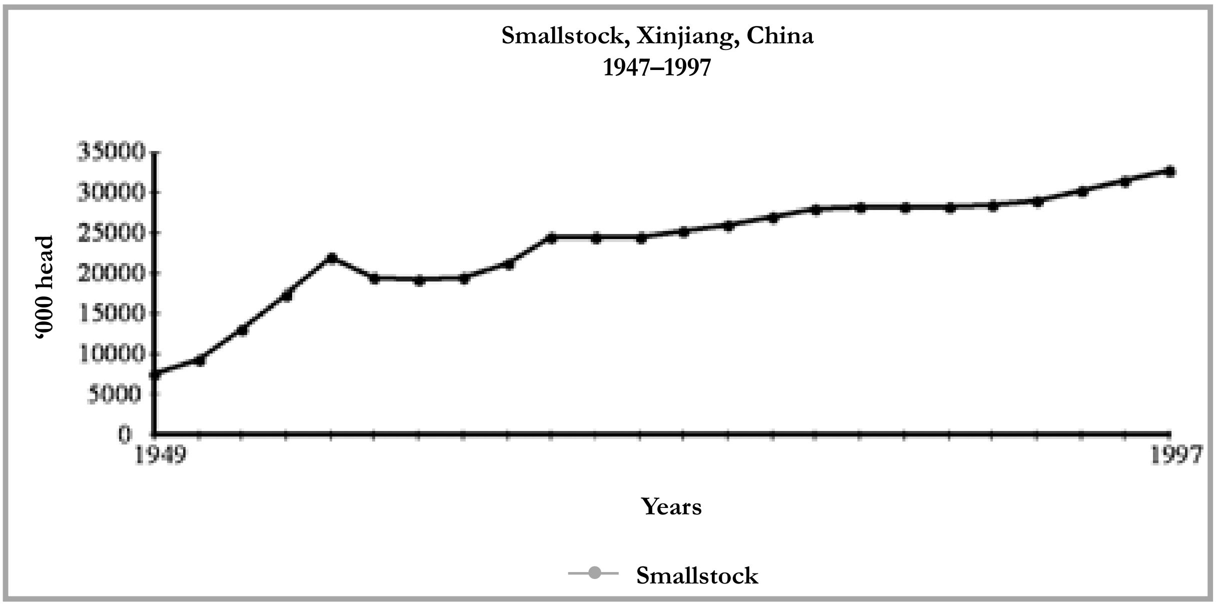
This discussion has shown that climatic variability and occasional climate-induced disasters have several implications for stocking rates, livestock mobility and property rights. Stocking rates need to be aimed as high as possible, if herders are to absorb the sudden loss of animals during climate disasters. Ideally, this loss of livestock can be turned into profit for herders, if good market outlets exist. Stable stocking rates cannot be maintained under variable and erratic climatic conditions. One of two options is required: bringing alternative sources of feed into the system, which however ultimately threatens rangeland sustainability, or allowing animals to move to alternative forage sources, which has become increasingly difficult to achieve.

VULNERABILITY OF LIVESTOCK AND RANGELANDS TO NON-EQUILIBRIUM ENVIRONMENTS IN NORTHERN ASIA

In sub-Saharan Africa, regular shortfalls in livestock grazing resources due to low rainfall have not been compensated by external feed supplies. The main pastoral responses to temporary livestock feed shortages have been to move livestock away from drought-afflicted areas and of migration of people from rangelands. When these options fail, pastoralists face famine. Rarely recorded in the past, such famines are now exposed to the global community and are not viewed as an acceptable outcome of climate patterns.

Problems stemming from ecosystem disequilibrium of rangelands in industrial parts of the world no longer have the same urgency. Perturbations due to climate have mainly been smoothed over by market mechanisms and provision of additional feed inputs. Climatic events in industrialised regions no longer leave large numbers of dead animals and dislocated people. In the 19th century cattle ranchers colonising the northern rangelands of the American great plains experienced huge livestock die-offs in droughts and blizzards (Young & Sparks 1985). Ranchers soon began to grow supplementary winter feed on irrigated land.

Figure 5: Livestock in Xinjiang province, China, 1949–1997 (Source: Statistical Office of Xinjiang 1997, cited in Hamann 1999)



In state-controlled rangeland livestock systems in the former Soviet Union, emergency climatic events were handled by transporting animals to other grazing areas or moving feed – in some cases by air lifts – to the animals. Feed resources external to the ecosystem were deployed on a massive scale to deal with temporary feed insufficiency due to these climatic events.

Pastoral vulnerability to climatic instability continued in those parts of Asia such as Tibet where the smoothing function of feed inputs had not been well established, as in the Soviet Union. In the past decade, large numbers of Asian pastoralists have again become susceptible to extreme climatic events, as it was up until 50 years ago before the implementation of large-scale industrialised and collectivised pastoralism. The extent of risk from droughts followed by winter snowstorms was demonstrated over four consecutive years from 1999 to 2002 in Mongolia, when millions of livestock died (Baas 2003). As yet, rangelands of the former Soviet republics have not experienced a major *dzhut* since the end of the USSR in 1991. It is essential to plan for this certainty.

In the political entities that cover northern Asia, pastoralists are now freed from the restraints of state control on livestock numbers. The possibility exists that rising livestock population pressure threatens the grazing resources. This is becoming an issue for example in Mongolia, where the goat population has doubled in the past decade. The non-equilibrium model suggests that without investment in supplementary feed, the growth of livestock populations would be restricted owing to the reduced quality and availability of forage over winter. Major climatic events would further reduce populations. These brakes on population growth would mean that grazing pressure, at least in the short term, would be unlikely to exceed pasture availability, except in localised areas. But privatisation of land for winter fodder production in China's rangelands has released livestock populations from this restraint. For some time now, Chinese scientists have warned that upward population trends of both humans and animals are causing excessive grazing pressure and degradation of the land (CID 1983; CSRPRC 1992; Banks et al. in press). The Chinese government has recently sought to impose restrictions on goat numbers.

A similar trajectory occurred under Soviet rangeland management in Kazakstan, as increasing stock levels were supported by supplementary feed while seasonal mobility was reduced, leading to degradation (Alimaev 2004; Ellis & Lee 2004). This trend is now in reverse. Small ruminant populations have fallen to 30% of their level at the end of the Soviet era (see Figure 4), and supplementary feed is no longer provided by the state. Some may argue that balance has been restored between pasture availability and livestock populations. However, this ignores the loss of peoples' livelihoods consequent upon such massive destocking, which also leaves large tracts of pastures completely ungrazed (Behnke 2004; Robinson et al. 2003). In this cold environment, some supplementary feed is necessary in winter, as the alternative of reverting to long distance migrations for winter pasturing is not open to most small-scale pastoralists (Kerven et al. 2003).

These contrasting trends under diverging political systems in north Asia leave several questions regarding the future. The first is whether an index can be developed for those rangelands in which temperature and precipitation variability both cause extreme swings in range plant biomass. In these regions, pastoral livelihoods are threatened if buffers do not exist against regular seasonal and unpredictable climatic events. Assessment of vulnerability is a first step towards devising protective and compensatory mechanisms. This is now recognized in very practical ways for example in Mongolia, through weather early-warning systems for pastoralists, group storage of emergency fodder supplies and an innovative livestock insurance scheme (Baas 2003).

The second question is whether appropriate policies can modulate between livestock numbers and the available feed resources under simultaneous conditions of ecosystem disequilibrium *and* rapid economic change in northern Asia. Where market forces are positive, as in China, livestock populations increase, to the despair of ecologists who warn of damage to the grazing resources. Direct state control of stocking rates, feed supplies and land use is no longer operable in the northern Asian rangelands. Yet laissez faire policies of letting the market and climate decide the fate of the rangelands – the reaction of most new Central Asian governments – is not a responsible option. More subtle policies, pricing instruments and remedial emergency measures are necessary to restore a balance between livestock populations, grazing resources and supplementary winter feed.

Discussants on African rangelands generally now oppose the notion that rangeland privatisation is necessary to maintain a safe balance between livestock populations and feed resources, under conditions of climate instability. However, little of this intense discussion has drifted across the continents to northern Asia, now in the midst of a rangeland privatisation experiment on a grand scale. Barbed wire fences are going up in China's rangelands. It is perhaps only a matter of time before the invisible fences used to regulate grazing in the state-controlled rangelands of the former Soviet Union are replaced by the real thing. Kazakstan's parliament, after years of debate, passed a new land law in 2003 that allows the sale of rangeland.

Researchers and users of rangelands in Africa and Asia have much to learn from each other. Parallel as well as divergent trends are apparent. The temptation is to

stretch the non-equilibrium model to fit all circumstances, but in doing so the model may become diluted and lose its explanatory power through broad generalisation. Efforts are now being made to sift through the data on northern Asia regarding the applicability or otherwise of the non-equilibrium model. Rather than seeking a global application of a model derived from one type of environment, our analytical tools need to be sharpened by re-examining a very different type of environment, using the insights learned from elsewhere.

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[1] Roy Behnke, Jim Ellis, Cees de Haan, Dan Miller, Wu Ning, Camille Richard, Euan Thomson and Nick Tyler provided very helpful contributions to an earlier draft of this paper.

[2] On the constraints of acquiring knowledge about Asian rangelands, it has been remarked how 'linguistically challenged scientists will find themselves frustrated when confronted by extensive Russian bibliographies published by obscure Soviet government agencies' (Thomas 1999:1048–52).

[3] See other papers in this volume.

[4] 717 million ha if including Arctic tundra, Siberian wetlands and other reindeer pastures (Vycius 1999).

[5] Across the Turkic languages of central and inner Asia, this term appears with slightly different stress; for example, *jut* in western China, *dzud* in Mongolia.

REFERENCES

Abel, N & Blaikie, P. 1989. *Land degradation, stocking rates and conservation policies for the communal rangelands of Botswana and Zimbabwe*. London: Overseas Development Institute. (Pastoral Development Network; no. 29a.)

Alimaev, I. 2004. Transhumant ecosystems: Fluctuations in season pasture productivity, in *Prospects for pastoralism in Kazakstan and Turkmenistan*: *From state farms to private flocks*, edited by C Kerven. London: RoutledgeCurzon.

Alimaev, I, Zhambakin, Z & Pryanishnikov, S. 1986. Rangeland farming in Kazakhstan. *Problems of Desert Development* (Ashgabat) 3:14–9.

Baas, S. 2003. *Designing a pastoral risk management strategy in Mongolia.* Paper presented at Commission on Nomadic Peoples session at International Union of Anthropological and Ethnological Sciences XV^th^ Congress, Florence 7–8 July. Rural Institutions and Participation Service, Food and Agricultural Organisation, Rome.

Babaev, A (ed). 1999. *Desert problems and desertification in central Asia.* Berlin: Springer.

Babaev, A, Nikolaev, V & Orlovskii, N. 1991. Current status and prospects for use of natural forage lands and dry farming in the Aral Sea Basin. *Problems of Desert Development*, 6:3–11.

Babaev, A & Orlovsky, N. 1985. Natural conditions of deserts in the USSR, in *Improvement of desert ranges in Soviet central Asia*, edited by N Nechaeva. Chur, Switzerland: Harwood Academic.

Batjargal, Z, Oyun, R, Togtokh, N & Sangidansranjav, S. 2002. Lessons learned from the Dzud 1999–2000, in *Fundamental issues affecting sustainability of the Mongolian Steppe*, edited by T Chuluun & D Ojima. Ulan Bataar: Mongolian Academy of Sciences/International Institute for Study of Nomadic Civilisation/Colorado State University. Interpress.

Banks, T. 1997. Pastoral land tenure reform and resource management in northern Xinjiang: A new institutional economics perspective. *Nomadic Peoples* (NS), 1(2):55–76.

Banks, T. 1999. State, community and common property in Xinjiang: Synergy or strife? *Development Policy Review*, 17(3):293–313.

Banks T, Richard, C, Li Ping & Yan Zhaoli. In press. Community-based grassland management in western china: Rationale, pilot project experience and policy implications.

Banskota, M, Karki, A & Croon, F (eds). 1999. *Strategic considerations on the development of central Asia.* Bussum, The Netherlands: Council for Sustainable Development of Central Asia.

Barfield, T. 1993. *The nomadic alternative.* Englewood Cliffs, New Jersey: Prentice-Hall.

Behnke, R. 2000. Equilibrium and non-equilibrium models of livestock population dynamics in pastoral Africa: Their relevance to Arctic grazing systems. *Rangifer*, 20(1):1–12.

Behnke, R. 2004. Reconfiguring property rights in livestock production systems of western Almaty Oblast, Kazakstan, in *Prospects for pastoralism in Kazakstan and Turkmenistan*: *From state farms to private flocks*, edited by C Kerven. London: RoutledgeCurzon.

Behnke, R & Abel, N. 1996. Revisited: The overstocking controversy in semi-arid Africa. *World Animal Review*, 87:4–27.

Behnke, R & Scoones, I. 1993. Rethinking range ecology: Implications for rangeland management in Africa, in *Range ecology at disequilibrium: New models of natural variability and pastoral adaptation in African savannas*, edited by R Behnke, I Scoones & C Kerven. London: Overseas Development Institute.

Behnke, R, Scoones, I & Kerven, C (eds). 1993. *Range ecology at disequilibrium: New models of natural variability and pastoral adaptation in African savannas*. London: Overseas Development Institute.

Bekenov, A, Grachev, I & Milner-Gulland, E. 1998. The ecology and management of the saiga antelope in Kazakhstan. *Mammal Review,* 28(1):1–52.

Benson, L & Svanberg, I. 1998. *China's last nomads: The history and culture of China's Kazaks*. New York: ME Sharpe.

Breman, H & De Wit, C. 1983. Rangeland productivity and exploitation in the Sahel. *Science*, 221:1341–7.

Central Asian Review. 1954. *Kazakstan: Animal Husbandry*, 2(1):85–96.

Channon, J & Channon, S. 1990. *The livestock sector in the republics of Soviet central Asia and Kazakhstan: A bibliographic study of works in Russian from 1917 to the present day.* Unpublished paper for Institute of Development Studies, Sussex University, Brighton.

Chuluun, T & Ojima, D. 1999. Climate and grazing sensitivity of the Mongolian rangeland system, in *People and rangelands: Building the future: Proceedings of the VI International Rangelands Congress, Townsville, Australia*, edited by D Eldridge & D Freudenberger:877–8.

CID (Consortium for International Development). 1983. *Range livestock production in the People's Republic of China*: Proceedings of the international symposium held at New Mexico State University, March 9–10, edited by B Stewart. Las Cruces, New Mexico: New Mexico State University.

Coppock, D, Swift, D & Ellis, J. 1986. Seasonal nutritional characteristics of livestock diets in a nomadic pastoral system. *Journal of Applied Ecology*, 23(2):585–96.

CSCPRC (Committee on Scholarly Communication with the People's Republic of China). 1992. *Grasslands and grassland sciences in northern China*. Washington DC: National Academy Press.

De Haan, C. 1994. *An overview of the World Bank's involvement in pastoral development.* London: Overseas Development Institute. (Pastoral Development Network; no. 36b.)

De Haan, C, Steinfeld, H & Blackburn, H. 1997. *Livestock and the environment: Finding a balance.* European Commission Directorate-General for Development, Development Policy Sustainable Development and Natural Resources, with Food and Agriculture Organisation, United States Agency for International Development and World Bank.

Delgado, C, Rosegrant, M, Steinfeld, H, Ehui, S & Courbois, C. 1999. *Livestock to 2020: The next food revolution.* International Food Policy Research Institute, Food and Agriculture Organisation of the United Nations, International Livestock Research Institute. www.ifpri.org/2020/briefs/number61.htm

Dmitriev, N & Ernst, L. 1989. *Animal genetic resources of the USSR.* Rome: Food and Agriculture Organisation. (Animal Production and Health Paper; no. 65.)

Dobie, J. 1980. *The Longhorns.* Austin: University of Texas Press.

Ellis, J. 1994a. Climate variability and complex ecosystem dynamics: Implications for pastoral development, in *Living with uncertainty: New directions in pastoral development in Africa*, edited by I Scoones. London: Intermediate Technology.

Ellis, J. 1994b. Ecosystem dynamics and economic development of African rangelands: Theory, ideology, events, and policy, in *Environment and agriculture: Rethinking development issues for the 21st century*, edited by S Breth. Morrilton, Arkansas: Winrock International:174–86.

Ellis, J & Chuluun, T. 1993. Cross-country survey of climate, ecology and land use among Mongolian pastoralists. Report to the Project on Alternatives for Livestock Development (PALD). Brighton: Institute for Development Studies, University of Sussex.

Ellis, J, Coughenour, M & Swift, D. 1993 Climate variability, ecosystem stability and the implications for range and livestock development, in *Range ecology at disequilibrium: New models of natural variability and pastoral adaptation in African savannas*, edited by R Behnke, I Scoones & C Kerven. London: Overseas Development Institute.

Ellis, J & Lee, R. 2004. Collapse of the Kazakstan livestock sector: A catastrophic convergence of ecological degradation, economic transition and climate change, in *Prospects for pastoralism in Kazakstan and Turkmenistan: From state farms to private flocks*, edited by C Kerven. London: RoutledgeCurzon.

Ellis, J & Swift, D. 1988. Stability of African pastoral systems: Alternate paradigms and implications for development. *Journal of Range Management*, 41:450–9.

Federovich, B. 1973. Natural conditions of arid zones of USSR and the ways in which livestock husbandry has developed in them, in *History of the economy of the peoples of Central Asia and Kazakhstan*, (TIE vol. XCVIII). Leningrad: Nauka Institute of Ethnography: 207–22. [in Russian].

Fernandez-Gimenez, M & Allen-Diaz, B. 1999. Testing a non-equilibrium model of rangeland vegetation dynamics in Mongolia. *Journal of Applied Ecology*, 36:871–85.

Fernandez-Gimenez, M. 1999. Reconsidering the role of absentee herd owners: A view from Mongolia. *Human Ecology*, 27(1):1–27.

Forbes, B. 1999. Land use and climate change on the Yamal Peninsula of north-west Siberia: Some ecological and socio-economic implications. *Polar Research*, 18(2):367–73.

Gilmanov, T. 1995. The state of rangeland resources in the newly-independent states of the former USSR, in *Rangelands in a sustainable biosphere: Proceedings of the Vth International Rangeland Congress*, vol 2, edited by N West. Denver, Colorado: Society for Range Management.

Gilmanov, T. 1996. Ecology of rangelands of Central Asia and modelling their primary productivity, in *Central Asia Livestock Regional Assessment Workshop.* Small Ruminant Collaborative Support Program and

International Center for Agriculture Research in Dry Areas. Tashkent 27 February–1 March. Davis: University of California.

Goldstein, M. No date. *Report on socio-economic constraints of herders*. Unpublished report for Qinghai Livestock Development Project, Qinghai Province, China.

Government of Tajikistan. 1999. *Agro-meteorological data 1965–92*. Compiled by K Qununov for Murghab station, Gorno Badakshan. Khorog, Tajikistan.

Hamann, B. 1999. *Kazakhs in Xinjiang, China*. Unpublished data from PhD thesis Ecological and socioeconomic impacts of the Chinese reform policy within the Southern border zone of the Dsungarian basin, Xinjiang, China. Landscape Planning Department, Technical University of Berlin.

Ho, P. 1996. *Ownership and control in Chinese rangeland management since Mao: the case of free-riding in Ningxia*. London: Overseas Development Institute. (Pastoral Development Network; no. 39c.)

Hodjakov, O. 1998. Personal communication. Institute of Animal Husbandry and Veterinary Services, Ashgabad, Turkmenistan.

Humphrey, C & Sneath, D. 1999. *The end of nomadism? Society, state and the environment in Inner Asia*. Durham, North Carolina: Duke University Press.

Illius, A & O'Connor, T. 1999. When is grazing a major determinant of rangeland condition and productivity? *People and rangelands: Building the future: Proceedings of the VI International Rangelands Congress, Townsville, Australia*, edited by D Eldridge & D Freudenberger:419–24.

Institute of Ethnography. 1973. *History of the economy of the peoples of central Asia and Kazakhstan*. Leningrad: Nauka [in Russian].

Kerven, C, Channon, J & Behnke, R. 1996. *Planning and policies on extensive livestock development in central Asia*. London: Overseas Development Institute. (Working paper; no. 91.)

Kerven, C. 1999. Unpublished field notes, Turkmenistan and Tajikistan, 1998–99.

Kerven, C, Alimaev, I, Behnke, R, Davidson, G, Franchois, L, Malmakov, N, Mathijs, E, Smailov, A, Temirobekov, S & Wright, I. 2003. *Retraction and expansion of flock mobility in Central Asia: Costs and consequences*. Paper for the VIIth International Rangelands Congress, Durban, South Africa.

Kerven, C (ed). 2004. *Prospects for pastoralism in Kazakstan and Turkmenistan: From state farms to private flocks*, edited by C Kerven. London: RoutledgeCurzon.

Khazanov, A. 1984. *Nomads and the outside world*. Cambridge: Cambridge University Press.

Krupnik, I. 1993. *Arctic adaptations: Native whalers and reindeer herders of northern Eurasia*. Hanover: University Press of New England.

Langvatn, R, Albon, S, Burkey, T & Clutton-Brock, T. 1996. Climate, plant phenology and variation in age of first reproduction in a temperate herbivore. *Journal of Animal Ecology*, 65:653–70.

Longworth, J & Williams, G. 1993. *China's pastoral region: Sheep and wool, minority nationalities, rangeland degradation and sustainable development*. Wallingford, Oxfordshire: CAB International.

Matley, I. 1989. Agricultural development, in *Central Asia: 120 years of Russian rule*, edited by E Allworth. Durham, North Carolina: Duke University.

McCarthy, N, Swallow, B, Kirk, M & Hazell, P (eds). 2000. *Property rights, risk and livestock development in Africa*. Washington DC: International Livestock Research Institute and International Food Policy Research Institute.

McMauley, M. 1994. Agriculture in central Asia and Kazakhstan in the 1980s, in *Political and economic trends in central Asia*, edited by S Akiner. London: British Academic Press.

Mearns, R. 1994. *Report on communal grazing issues*. For ULG Consultants and Ministry of Agriculture, Government of Kyrgyzstan, Bishkek.

Miller, D. 1997. New perspectives on range management and pastoralism and their implications for HKH-Tibetan plateau rangelands, in *Rangelands and pastoral development in the HKH* [Hindu Kush Himalaya]. Proceedings of a Regional Experts' Meeting, edited by D Miller & S Craig. International Centre for Integrated Mountain Development, Kathmandu.

Miller, D. Forthcoming. Snowstorms, restocking and settling nomads on the Tibetan Plateau in Western China, in *Livestock and livelihoods: Current perspectives on restocking*, edited by C Heffernan. Reading: Reading University.

Miller, D. 1999. Herders of forty centuries: Nomads of Tibetan rangelands in western China, in *People and rangelands: Building the future: Proceedings of the VI International Rangelands Congress, Townsville, Australia*, edited by D Eldridge & D Freudenberger 402–3.

Milner-Gulland, E. 1994. Sustainable management of the saiga antelope. *Oryx*, 28(4):257–62.

Nechaeva, N (ed). 1985. *Improvement of desert ranges in Soviet central Asia*. Chur, Switzerland: Harwood Academic Press.

Neupert, R. 1999. Population, nomadic pastoralism and the environment in the Mongolian plateau. *Population and Environment*, 20(5):413–41.

Niamir-Fuller, M (ed). 1999. *Managing mobility in African rangelands: The legitimization of transhumance*. London: Intermediate Technology.

Nicholls, N & Wong, K. 1990. Dependence of rainfall variability on mean rainfall, latitude, and the southern oscillation. *Journal of Climate*, 3:163–70.

Olcott, M. 1981. The settlement of the Kazakh nomads. *Nomadic Peoples*, 8:12–23.

Olcott, M. 1995. *The Kazakhs*. 2nd edition. Stanford: Hoover Institution Press.

Purev, B. 1990. Traditional pastoral livestock management in Mongolia. In *Proceedings of International Workshop on Pastoralism and Socio-economic development. Mongolia (4–12 September)*. Food and Agriculture Organisation, Rome.

Reynolds, S, Batello, C & Baas, S. 1999. Perspectives on rangeland development: The Food and Agriculture Organisation of the United Nations (FAO). *People and rangelands: Building the future: Proceedings of the VI International Rangelands Congress, Townsville, Australia*, edited by D Eldridge & D Freudenberger:160–5.

Richard, C & Miller, D. 1998. Grazing commons of the Hindu-Kush Himalaya and Tibetan Plateau: Constraint or opportunity? 1999. *Strategic considerations on the development of central Asia*. Bussum, The Netherlands: Council for Sustainable Development of Central Asia.

Robinson, S. 2000. *Pastoralism and land degradation in Kazakhstan*. Unpublished PhD thesis, Department of Biological Sciences, Warwick University.

Robinson, S, Milner-Gulland, E & Alimaev, I. 2003. Rangeland degradation in Kazakhstan during the Soviet era: Re-examining the evidence. *Journal of Arid Environments*, 53:419–39.

Rozelle, S, Huang, J & Zhang L. 1997. Poverty, population and environmental degradation in China. *Food Policy*, 32(3):229–51.

Ryder, M. 1983. *Sheep and man*. London: Duckworth.

Scoones, I (ed). 1994. *Living with uncertainty: New directions in pastoral development in Africa*. London: Intermediate Technology.

Seligman, N & Perevolotsky, A. 1994. Has intensive grazing by domestic livestock degraded the Old World Mediterranean rangelands? In *Plant-animal interactions in Mediterranean-type ecosystems*, edited by M Arianoutsou & R Groves. Dordrecht: Kluwer.

Sneath, D. 1998. State policy and pasture degradation in inner Asia. *Science,* 281:1147–8.

Sneath, D. 1999. Spatial mobility and inner Asian pastoralism, in *The end of nomadism? Society, state and the environment in inner Asia*, edited by C Humphrey & D Sneath. Durham, North Carolina: Duke University Press.

Thomas, I. 1999. The recent history of rangelands in western Asia, in *People and rangelands: Building the future: Proceedings of the VI International Rangelands Congress, Townsville, Australia*, edited by D Eldridge & D Freudenberger:1048–52.

Thwaites, R, DeLacy, T, Hong, T & Hua, L. 1998. Property rights, social change and grassland degradation in Xilingol Biosphere Reserve, Inner Mongolia, China. *Society and Natural Resources* 11(4):319–38.

Todes, D. 1989. *Darwin without Malthus: The struggle for existence in Russian evolutionary thought*. New York: Oxford University Press.

Tserendulam, R. 1990. Natural pasture as the basis for livestock production in Mongolia, in *Proceedings of International Workshop on Pastoralism and Socio-economic Development. Mongolia (4–12 September)*. Rome: Food and Agriculture Organisation.

Van Veen, T. 1995. *The Kyrgyz sheep herders at a cross roads*. London: Overseas Development Institute. (Pastoral Development Network; no. 38d.)

Vitebsky, P. 1990. Centralised decentralisation: The ethnography of remote reindeer herders under Perestoika. *Cahier du Monde Russe et Soviétique*, 31 (2–3):345–56.

Vycius, J. 1999. Rangeland resources on the former Soviet Union territory. *People and rangelands: Building the future: Proceedings of the VI International Rangelands Congress, Townsville, Australia*, edited by D Eldridge & D Freudenberger:796–7.

Webster, A. 1993. Nutrition and the thermal environment, in *Nutritional physiology of farm animals*, edited by J Rook & P Thomas. London: Longman.

Williams, D. 1996. Grassland enclosures: Catalyst of land degradation in Inner Mongolia. *Human Organisation*, 55(3):307–13.

Wu Ning. 1997. Rangeland resources and conditions in Western Sichuan, in *Rangelands and Pastoral Development in the HKH* [Hindu Kush Himalaya]. Proceedings of a Regional Experts' Meeting, edited by D Miller & S Craig. International Centre for Integrated Mountain Development, Kathmandu:23–40.

Wu Ning and Richard, C. 1999. The privatisation process of rangeland and its impacts on the pastoral dynamics in the Hindu Kush Himalaya: The case of Western Sichuan, China, in *People and rangelands: Building the future: Proceedings of the VI International Rangelands Congress, Townsville, Australia*, edited by D Eldridge & D Freudenberger.

Xinshi, Z. 1992. *Northern China: Grasslands and Grassland Sciences in Northern China*. Committee on Scholarly Communication with the People's Republic of China. Washington, DC: National Academy Press:39–54.

Young, J & Sparks, B. 1985. *Cattle in the cold desert*. Logan, Utah: Utah State University Press.

FACTORS AFFECTING THE ECONOMIC ASSESSMENT OF OPPORTUNISTIC AND CONSERVATIVE STOCKING STRATEGIES IN AFRICAN LIVESTOCK SYSTEMS

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INTRODUCTION: PURPOSE, DEFINITIONS AND HISTORY

The purpose of this paper is to review some issues in the *economic* assessment of livestock systems in Africa, and in particular the assessment of the relative superiority of strategies affecting the stocking rate. I appreciate that 'setting the stocking rate' is only one of the tools in the management of range and animals (Tainton 1999, especially Chapters 7 & 12), but it is one which has pre-occupied academic ecologists, economists and other social scientists as well as policy-makers. It is, perhaps, the issue which has done most to unite the social scientists and the ecologists in the so-called 'new rangelands science', that is, the adherents of the non-equilibrium paradigm, although their reasons for unification are more tactical (here is an argument that supports my point of view!) rather than theoretical.

The trigger for this paper was an article by Campbell and his associates (Campbell et al. 2000)[1] (hereinafter 'CEA') which criticised the 'new rangeland science' for its belief that opportunistic strategies give higher economic returns compared to strategies based on conservative stocking strategies; and purported to show that the evidence available, when simulated to represent a livestock system over 15 years, indicated that a conservative stocking policy was economically superior. This prompted a revisit and re-thinking of the issues involved. CEA particularly attacked the new range scientists for their neglect of the capital costs, in terms of the original purchase and subsequent restocking of animals after drought.

In this debate 'opportunism' and 'conservatism' are normally defined in terms of stocking rates per hectare. At its most basic, a 'conservative' strategy is one which, through a combination (mainly) of decisions on sales and purchases of animals, ensures that the actual stocking rate (in terms of animals or grazing liveweight per hectare) is kept constant at a level such that the animals never (or very seldom) suffer a shortage of the feed they need (for example during a drought). Such feed shortages are most likely to occur because of fluctuations in feed supply caused mainly by fluctuations in rainfall. A conservative strategy will (most probably) result in there being surplus feed supply unused, because there are too few animals to eat it, at other non-drought times when the feed supply is greater than the feed demanded by this conservative level of livestock numbers. In contrast, an opportunistic strategy is one which, by the same sort of decisions as in the conservative case, allows the number of animals to fluctuate continuously in order to balance exactly the demand for a supply of livestock feed. This is also termed a 'tracking' or 'tight-tracking' strategy.

Of course more sophisticated definitions and decision mechanisms are possible. For example, the demand and supply can be restated in terms not of 'needs' but of 'appetite' while the available supply may be restated in terms of the amount which can be safely harvested without ecological degradation. Different definitions lead to changes in the precise number of animals that can be kept. But, provided the definition currently being used is clear, these different definitions do not substantially change the nature of the issue. We shall stick with the most simple here. Illius et al. (1998) provide examples of how particular decision criteria (for example, on the age-and-sex categories of animals purchased or sold) for stocking, de-stocking and restocking, in the general context of opportunism and conservatism can lead to different results.

The issue is a very long-standing one, and it should be stressed that variation in stocking rate is only one of the

strategies used by pastoralists and settled farmers in semi-arid areas to combat the effects of climatic uncertainty on the productivity of their animals. Other important strategies are mobility, a concentration on drought-tolerant breeds (for example, Zebu cattle) or species (such as camels), or alternatively a diverse mix of livestock species, and general income diversification (Niamir-Fuller & Turner 1999). The terms 'opportunistic' and 'conservative' began to be used in the context of this issue in the late 1970s, and two pieces by Sandford (1982 & 1983), pointing out some advantages of an opportunistic strategy and the factors that would affect its desirability, are frequently referred to as the origins of recent multi-disciplinary professional discussion.

In the mid-1980s De Ridder and Wagenaar (1984) drew attention to the importance of including all the outputs of a pastoral system when comparing the productivity of 'traditional' (which usually favour an opportunistic strategy) and 'commercial ranching' systems (which often claim they pursue a conservative strategy). The inclusion of other outputs, rather than just meat/live-animal sales, usually suggests the higher output per hectare of traditional rather than ranching systems and, by a somewhat perilous extension, of the stocking strategies they pursue. The argument received fresh impetus with the publication of the proceedings of two international workshops held in 1990 and 1993 (Behnke et al. 1993; Scoones 1994), which tended to endorse opportunism. In the African context a number of field studies, for example, Scoones (1990) and Barrett (1992) both based on Zimbabwe, provided survey evidence of the higher output of African semi-arid livestock systems in contrast to estimated figures of output from commercial ranches. Later other work, for example, Illius et al. (1998) and Campbell et al. (2000), started to do detailed calculations of the merits of opportunism and conservatism in particular context, with varying results (discussed below).

The studies quoted above were all done in an African context, which is where the expressions 'opportunism' and 'conservatism' are most frequently used and the debate liveliest. The discussion there is often couched in policy or strategic terms involving human communities or general types of livestock-keeping enterprises. The same basic issue (management of stocking levels) is also tackled, in a slightly wider context of rangelands management strategies, in the USA and Australia. In their cases the focus is often on the individual enterprise and takes the form of software programmes able to provide profit-increasing advice, with an increasing emphasis on the use of dynamic programming techniques (see, for example, Rodriguez & Roath 1987; Carande et al. 1995; Batabyal et al. 2001; Sugiharto & Pannell 1998). Dynamic programming has also been used at least once for analysing African pastoral systems (Mace & Houston 1989).

ISSUES IN THE ECONOMIC ANALYSIS

There are a number of significant issues in designing an appropriate economic analysis of stocking strategies

WHICH GOODS AND COMMODITIES?

The first of these concerns the commodities/goods that are to be included in the analysis. Some evaluators carry out the analysis in terms of only one commodity, for example, Brown 1971 (milk), Illius et al. 1998 (liveweight), implicitly or explicitly assuming either that it is the only good of real value or that the decisions and biological processes involved in the production of that good will always lead to the production of other goods in identical proportions (in value terms, to each other) so that reiteration of the calculation to include each commodity will not change the order or desirability indicated by the calculation in respect of the single good.

Neither assumption is valid. Given, for example, that the relative price of milk and meat vary from time to time and place to place, that the ratio between the quantity of outputs of different goods (say milk and meat) can be varied, for example, by decisions about the age and sex composition of the herd, there can be no reliance on the constancy in the ratio of the value of output between different goods. Nor is one product, for example, liveweight/meat, universally predominant so that one can ignore the other goods as being of negligible importance.

Livestock in African systems produce a wide range of goods. This applies both to 'purer' pastoralists (such as the Borana or Somali in East Africa) and to settled farmers in semi-arid areas of Zimbabwe. Coppock (1994) found that among pastoral Borana in Ethiopia, 90% of total household income came from cattle, less than 1% of cattle output was an input (for example, draught power or manure to cropping activities), and that milk (marketed or consumed) accounted for 40% of livestock output. Among camel herders up to 150 kilometres from the capital of Somalia, Heeren (1990) found in the late 1980s, that 50–60% of pastoralists sold camel milk and the majority of these earned more than half their cash income from this source.

Campbell et al. (2000) quote the figures of relative proportions of different commodities in total livestock production, which are set out in Table 1 for various parts and sources in Zimbabwe. The figures show considerable variations in proportions both between commodities for a single source/area and between sources/areas for a single commodity. In most but not all studies in Zimbabwe, the biggest contributor to total value of output is animal traction (both ploughing and transport). It is, coincidentally, the area about which we have the least knowledge – especially about the physical and economic determinants of supply and demand.

The evidence presented in Table 1 should inhibit anyone from believing that an economic appraisal of the relative economics of one output only is adequate to understand the economics of the whole livestock system in different parts either of Zimbabwe or of other African countries.[2]

A more contentious point arises in respect of the inclusion of range degradation, which may take the form of a change in vegetation cover, vigour and composition (see section by Morris et al. in Tainton 1999:186–93) and of soil erosion, and consequent loss of soil depth and nutritional quality, especially organic matter (Snyman 1999). Should the loss of, or increased variability in output, consequent on range degradation, be taken as another 'negative good' to be taken into account as a cost in the calculations? Theoretically it should, both from a general social point of view and also from a private one if the land-user is also the landowner. In practice, few economic evaluations take quantitative account of land degradation. The figures are difficult to obtain and they vary enormously according to soil, slope, rainfall intensity, aridity, and other factors. Work by Biot (1993) and Abel (1993) has suggested that serious loss of productivity due to overgrazing may take centuries rather than decades. In spite of the caveats by these authors that their findings are specific to a particular ecosystem, evaluators may have been lured into indolence by the impression that (due to the time-discounting factor used by economists – see next sub-section of this paper) these effects of overgrazing will have little impact on economic evaluations.

WHAT ECONOMIC CRITERION?

The second point of substance concerns the economic criteria according to which opportunistic and conservative strategies and their variants are to be valued and ranked. Following a lead set by Herskovits (1926), many students of livestock systems in Africa have believed that African pastoralists are not really motivated by 'economic' criteria at all and that, at best, the application of economic criteria will be constantly at odds with the non-economic motivation of African producers so that economic incentives to improve performance will be useless. That train of thought seems to be dying out. This is partly because the allegedly non-economic aim of maximising cattle numbers is now accepted as being just as 'economic' as the desire to maximise other forms of income, wealth or 'utility'. It is also because cattle-accumulation activities which previously could not be explained in other ways, are now seen, as a result of a better appreciation of cattle dynamics and productivity (the seminal works on this issue are Brown 1971 and Dahl & Hjort 1976), as being adequately explicable in terms of the same income, wealth or utility-maximisation criteria as are thought to be pursued by non-pastoralists.

In spite of western economics' adherence to the principle that the 'consumer is king', I am unaware of any serious field study among African pastoralists to elucidate which of these economic criteria their own decision models adhere to most closely. Some conceptual advance has been made in papers by Mace (1988) and Borgerhoff et al. (1994), but it has not, apparently, resulted in the sort of specific quantitative criteria normally used by economists. Conventional economic criteria applicable elsewhere are therefore applied indiscriminately to the evaluation of pastoralists' strategies and those of settled farmers in semi-arid areas; and can be criticised or defended in conventional economic terms.

Three criteria are most commonly used. One of these is gross or net (of recurrent costs) output per unit of the scarcest resource (conventionally taken as land, but in particular circumstances it could be a person or unit of water). A result would typically be expressed as an annual average number of dollars per hectare. That is probably the most common method used in evaluation studies (see comments on this by Campbell et al. 2000:414, 430) but it is, from an economist's point of view, theoretically indefensible as it ignores the cost of capital involved in establishing the enterprise (that is, ranch or family herd). It could be defended on practical grounds if capital costs are negligible (which is not the case with livestock enterprises), or if the capital investment concerned can be considered as a sunk cost, that is, one which has already been incurred and can not be recouped by sale as

Table 1: Proportions of gross output contributed by various commodity outputs in Zimbabwe

SYSTEM AND SOURCE	PLOUGHING	TRANSPORT	MANURE	MILK	SALES AND SLAUGHTER
Chivi, (survey and subsequent simulation by Campbell et al. 2000)	7%	25%	11%	24%	32%
20 veld-management schemes in Masvingo (Danckwerts n.d.)	41%	7%	52%		
Communal area cattle (Barrett 1992)	63%	9%	14%	14%	
Chivi (Scoones 1990)	56%	16%	3%	23%	2%

a going enterprise or by selling off the stock. This second defence might have been credible 40 years ago in east Africa, but the increased rate of migration from pastoral to non-pastoral occupations, and the increasing incorporation of pastoral enterprises in market operations makes it incredible now.

The second commonly used criterion is expressed in profitability terms, either as annual average profit from livestock production per hectare, or as a rate of return on capital. It is the criterion most often used by professional accountants, and we shall call it, for short, here 'the accountant's criterion'. The criterion averages annual (or some other period, but annual is the most common) *gross* output (sold or consumed) of all the relevant goods, that is, live animals, on-farm slaughters, milk, ploughing and transport services and then deducts annual average current livestock expenditure (for example, on feed or veterinary expenditure, and sometimes the opportunity cost of herding and watering labour, but excluding expenditure on purchasing animals). The number of years over which the average is taken may be either the life of the enterprise, if this is finite, or a sufficient number of years to be representative of climatic conditions. This gives a value of annual average *net* output per hectare.

To calculate profitability, the net output figure must then be adjusted to allow for capital costs. These can be calculated on an annual basis by adding the cost of the herd (the original cost plus the cost of any purchased replacements) and then deducting from this figure the value of the herd at the end of the actual or predicted life of the enterprise. The resulting total is then divided by the number of years involved in the calculation of the benefits. No depreciation charge is made for the capital costs of the livestock since 'replacement' takes place primarily through in-herd births and deaths and any imbalance between these two is reflected in the 'end-value' of the herd. The average annual capital cost calculated in this way is then deducted from the average net output per hectare to give a figure for 'profit per hectare'.

Economists tend to be scornful of the accountant's criterion, based as it is on annual averages, because it takes no account of the distribution over time of the costs and outputs (benefits) involved. Economists emphasise that the earlier a cost is incurred the sooner resources have to be freed from consumption or alternative investments in order to fund this cost. Similarly the sooner an output or benefit is produced the sooner can consumption be increased or an alternative profitable investment undertaken. They, therefore, use a discount rate to reduce costs and benefits incurred at different times to a common denominator, expressed as a present value, by discounting. At a 10% discount rate, for example, a dollar of expenditure is valued as $1 if it occurs in the present year (year 0), but has a present value of only $0.62 if it occurs in year 5 (that is, 1 cumulatively discounted by 10% 5 times) or 0.26 if it occurs in year 15. The discount rate measures (either or both) the extent to which consumers prefer present to future satisfaction (if you do not believe they do then try fobbing off a child with the promise of an ice-cream tomorrow rather than today!) or the rate of return that can be earned on an alternative investment. This is a complex subject, one which few economists really understand and fewer still are able to explain to non-economists. As clear a short explanation as I have seen occurs in the technical appendix to a World Bank handbook (Belli et al. 1997:122).

While there are different techniques to use 'present values', as defined in the previous paragraph, in measuring economic performance, the one which economists tend to prefer is one called the 'Net Present Value' (NPV) of a project. The present values of a project's benefits and costs in each year of its life are summed and if the present value of the benefits exceeds the costs, the project is 'economically desirable'. If the costs exceed the benefits it is undesirable. The concept of a project, in this context, can be extended to a strategy such as stocking opportunism or conservatism in which large numbers of people act in the same or similar ways. It is an assumption underlying the NPV criterion that all inputs and outputs can be purchased or otherwise acquired at some price except for a single aspect, for example, a hydro-electric site, which defines that project because it can not be simultaneously used by another project. Where two or more projects compete for a site the project with the highest NPV (if positive) should be undertaken. Where two strategies (for example, opportunism and conservatism) are incompatible (that is, they can not be practised simultaneously on the same piece of rangelands), then the one with the higher NPV should be undertaken. CEA have used this technique in their comparison of conservative and opportunistic strategies among sedentary farmers with livestock in Zimbabwe.

The strength of the NPV criterion (that it takes account of the time at which benefits and costs occur) is also its weakness. It is a strength, for example, because the capital cost of purchasing animals primarily occurs early in project/strategy life, and these initial set-up costs can be reduced by purchasing young animals but only at the expense of postponing much output until these animals have had time to grow, reproduce and lactate. NPV calculations provide a way of evaluating these contrasts in timing, which the accountant's criterion does not. However this strength is also its weakness. Livestock outputs, calving, mortality and the need for restocking are all dependent on the availability of livestock feed, which is itself predominately dependent on rainfall. Rainfall is highly variable in range areas. It is also unpredictable. The consequence is that a strategy that

may have been preferable to another if the project starts in one year may be inferior to its rival if the project starts in another year.

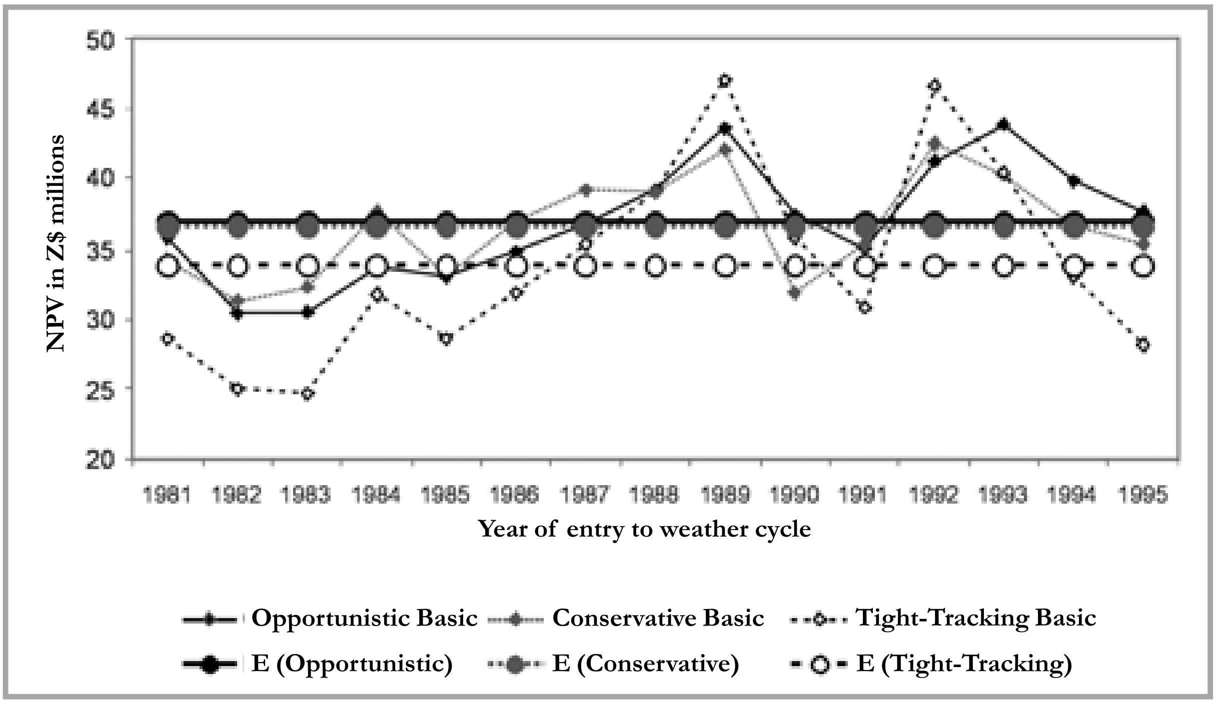
Figure 1 illustrates this. It is drawn from the results of a computerised spreadsheet model which mimicked, as far as this could be done,[3] the results of CEA's economic analysis, but altered one key assumption and several technical coefficients where these were thought to be unrealistic. It shows how the NPV of three stocking strategies vary, both absolutely and in relation to each other, depending on the year in which one enters a cycle of years based on actual rainfall in Chivi in 1981–1995.

The thin lines (punctuated by diamonds) in Figure 1 trace changes in the NPVs of three alternative strategies (or scenarios) that occur as the year in which one enters the weather cycle changes from 1981–1995 to 1982 (via 1995)–1981, and so on. As one can see in the figure, the Opportunistic Basic scenario (equivalent to what farmers actually do, which is a modified form of opportunism) is preferable if one enters the weather cycle in 1981, 1988, 1990, 1993 or 1994. The Conservative Basic scenario is preferable if one enters the weather cycle in 1982–1987 or 1991, and the Tight Tracking Basic (which is equivalent to the Opportunistic as defined in the present paper) is superior if one enters the weather cycle in 1989 or 1992. The weather is unpredictable, however, so that even if one thinks there is a weather cycle and knows roughly where one is in relation to it, one can not be certain enough to be sure of a successful outcome.[4]

Economists have an answer to this in their concept and calculation of the 'expected value of the NPV' of a project. In essence the argument is that if you always rank projects by their NPV even though you will get it wrong in a number of cases, overall you will come out better off than if you followed another investment rule. Dorfman (1962:129) wrote 'In design and operating decisions the results of which are influenced by chance or unknown factors such as the whims of weather this simplification (that is, "it is frequently expedient to ignore uncertainty") is clearly untenable'. While exploring and recognising the danger of relying on 'expected values' as a decision criterion (in simple form they ignore risk aversion), Dorfman (1962) recommended that the calculation of the *expected* value of NPV should be an essential step in the analysis. The expected value of a variable can be defined as the sum of all the possible values that the variable can take, with each value being multiplied by the probability of its occurrence. The sum of the probabilities is 1. Following Dorfman's advice (although his caveats are often ignored), the use of 'expected values' is now fairly common in investment appraisal.

In Figure 1 the thick level lines (punctuated by solid circles) represent my best guess of the expected values of the NPVs of the three strategies (scenarios) represented in that figure, which have already been discussed. They are approximations to the expected values, and are obtained by averaging the NPVs over 15 simulations. A more correct way would have been to calculate the expected values of the benefits and costs separately, taking into account any covariance in their values.

Figure 1: Variations in value of NPV of different stocking strategies, by scenario and by year of entry to weather cycle (8% discount rate)



As far as I know, no published evaluation of the performance of different stocking strategies in Africa has made use of the 'expected value' of the NPV criterion. Theoretically it has attractions. In practice there are difficulties in computation but these, possibly, are not so difficult as persuading yourself and others that a technique, which is a bit of a black box even to economists, is an entirely rational approach to economic evaluation. In the light of the evidence presented in Figure 1, which shows the changes in the relative performance of different strategies depending on the year of entry into a weather cycle, clearly an NPV based on the evidence of a single run of years is unsatisfactory also. The accountant's average profit criterion has a certain simplicity to recommend it. However it systematically overvalues the performance of strategies with high initial costs and low initial benefits compared to alternative strategies.

THE COMPLEXITY OF THE ANALYSIS

THE NECESSITY FOR SIMULATION

Livestock systems, particularly cattle systems, react slowly compared to ones involving annual crops. This is because of the low fecundity and low rate of growth of livestock populations and hence the delayed reaction of output to earlier change in management and inputs. A change, for example, in conception rate in Year$_t$ will have some effect on milk output in Year$_{t+1}$, but it will also have a more substantial effect in Year$_{t+4}$ when the increased calves born in Year$_t$ will themselves calve and lactate. The consequence is that, in areas of high inter-annual variation of rainfall and therefore of feed availability and animal productivity also, a large number of observations are necessary before you can have a reasonably reliable estimate of economic yield or other parameters.

It is difficult enough to carry out biological or physical experiments or surveys over long periods of years. It is frankly impossible to carry out economic surveys over long periods. Your phenomena migrate elsewhere, they change strategies (to some extent), they get bored and refuse to be observed, or, worst of all, they get devious and deliberately corrupt the observations by concealing or inventing outputs and costs.

The consequence is that you cannot, through conventional or experimental techniques which directly measure the net profitability of many livestock enterprises over a large number of years, generate sufficient data to give reliable or stable estimates of output, inputs and hence the profitability of different strategies. You have little alternative to devising a simulation model that will use a causal structure dependent on one or a few causal factors (for example, mainly rainfall) about whose statistical incidence you have some knowledge to generate a distribution of the NPVs.

ECONOMIC ANALYSIS BY SIMULATION MODEL

I am not a specialist in building models but I have been involved in policy analysis in African rangelands for several decades. Since model builders frequently claim that their models can be useful for policy analysis-and-making I am in a position to comment as a potential user of such assistance.

There is a large number of biological models simulating vegetation and animal performance in rangelands. The most sophisticated of them were designed for use in North America and Australia. For example, *GRAZPLAN* (and its components), a simulation package which can be purchased commercially in Australia, is a short-term model which calculates the output from a herd or flock of three different outputs, milk, live-growth and wool. To do this involves 132 estimating equations (Freer et al. 2002), and requires the user to provide site/time-specific information on about 30–40 parameters. This model does not incorporate any economic data or processes.

In this section I shall briefly mention two simulation models that have been used to throw light on the advantages of alternative stocking rates in southern Africa. These two are based on both experimental and survey data and tend to go into considerable detail in simulating biophysical processes.

The most substantial effort has been that by the Rangeland Modelling Group attached to the Department of Mathematics and Applied Mathematics at the University of Cape Town. Over the last 12 years they have produced between 30 and 40 papers (the most recent of which is Richardson et al. 2000) on various aspects of modelling range and livestock production systems. Some of these are detailed mechanistic models of particular aspects of plant or animal production. Others integrate these production system models into simulations of entire rangelands.

One of these papers, Richardson et al. (2000) points out the practical impossibility of using very detailed mechanistic models and integrating their results, and have tried to limit the amount of information required and used by devising simpler phenomenological models, essentially summarising the results of more detailed mechanistic models and incorporating these in a framework model which brings together the plant, animal intake and animal production sub-models. However in this paper they still end up with a simulation model with 44 differential equations to cover two commodities (live sales and milk), and it is entirely a biophysical model without any economic component, that is, without any monetary values or prices. It is not clear to what extent there is feedback from reproduction to numbers in subsequent periods. The model they describe in the paper is one applicable to the semi-arid

savanna, and they report that they need to develop another one for a different environment, that is, the arid shrub lands of the Karoo. In other words their models have only limited generalisability.

A paper and model that directly addresses the opportunism versus conservatism issue is that of Illius et al. (1998 with subsequent corrigendum). They simulated the performance of animal production in semi-arid savanna over 20 years under eight management policies (strategies). Each of these strategies applied particular rules about stocking, destocking and restocking. Particularly interesting among their findings are:

- The strong positive effect of stocking rate on per hectare and system output. The relative performance of different strategies was highly dependent on the actual stocking rate they achieved.
- However the stocking rate actually achieved in the simulation was not always the one intended. What was actually achieved depended very much on the precise rules used to implement the strategy, particularly in respect of the timing (during the year) of stocking and destocking, as well as which class of stock were involved in the destocking.
- The physical results of the simulation were strongly affected, through the processes of mortality and reproduction, by the precise order in which the years of randomised rainfall records were fed into the simulation. This is a parallel to, but not the same problem as, the one that affects the calculation of the NPV and for which the calculation of expected values is the economist's solution.

The third bio-physical study of whose results I shall make use below is by Fynn and O'Connor (2000). It is not a simulation model, but presents the results of a ten year experiment at two sites, involving different stocking rates. By using replaceable/replaced Brahman weaners it avoided the complications caused by reproduction.

Common characteristics of these biophysical studies and models are:

- They are enormously data-intensive;
- Their authors insist on their limited generalisability outside the particular ecosystems for which they were designed;
- They include only some of the economically important outputs.

Whatever their long-run future, it seems improbable to me that, if each ecosystem needs its own model, they will provide widely applicable policy conclusions in the next 5–10 years. This is particularly the case outside South Africa.

In marked contrast to these exclusively biophysical simulation models is the largely economic one developed by CEA, which Ian Scoones and I have tried to mimic (Sandford & Scoones 2003). I call this the 'CEA-mimic model' and have had to describe it, (rather than the CEA original here) because we have not had access to the original's software. I have done some further work along similar lines, which I refer to (rather grandiosely) as the 'Sandford' model. The original CEA model was written as a spreadsheet model, as are the mimic and Sandford models.

Following CEA, the CEA mimic model tried to simplify the biological component of an integrated five-commodity biological-cum-economic model of the livestock component of mixed farmers. The model is basically driven by rainfall as moderated by assumed technical and price coefficients and by the rules (for stocking rates, sales, and purchases) applicable to each scenario. It was developed for application to two areas in Zimbabwe. We shall concentrate on the results for one, Chivi, in the semi-arid zone. The 'CEA-mimic' model has the following specifications set out below, which are likely to be roughly matched in the CEA model. I have slightly simplified the description by omitting some sub-routines.

For each of four alternative scenarios (two termed 'conservative' and two 'opportunistic') the mimic model involved:[5]

- One equation relating annual feed availability to annual rainfall;
- One equation relating the total number of animals kept in a year to feed availability in the current year;
- One equation determining, on the basis of feed availability, the number of animals (after allowing for last year's balance of stock plus current births and mortality less replacements) to be retained in the herd, the number of animals taken off (slaughtered and sold) and the number of animals purchased as replacements.
- One equation relating grazing pressure to feed availability in the current and previous year (mimic model, not in CEA original)
- Five equations quantifying the annual quantity of commodities produced, of which there are five, that is, milk (home-consumed or sold), animals sold or slaughtered, manure, ploughing power and transport services. The total quantity of each good produced is determined by one or more of the following factors (depending on the commodity):
  a) The total number of animals kept
  b) A coefficient specifying the proportion of animals in the herd producing that commodity. The size of the coefficient is either fixed or varies with rainfall or grazing pressure.
  c) A yield coefficient, either fixed or related to current rainfall or grazing pressure.
- Five equations relating the prices of the five commodities to the quantity produced or marketed (net offtake after allowing for purchases in the case of animals and meat).

The above equations essentially determine the total value of the gross output from the herd in each year of a 15-year simulation. 15 years are chosen because that was the number of years of data available to CEA.

In addition there are three equations determining the annual level of three kinds of cost (one equation each for the cost of livestock, for other capital expenses (some fencing), and for recurrent costs.

Subsequent to reaching that point there are a number of simple subroutines discounting the sums of costs and outputs in each year to give an NPV (net present value) for each scenario in each of the two strategies. There is also (in the mimic but not in the original CEA version) a slightly more complex macro reiterating each 15-year weather cycle, but starting one year later in the cycle. This enables the calculation of an 'expected value'.

In the CEA original and in its mimic the differences between the scenarios/strategies are accounted for by relatively fixed (over time) coefficients favouring the conservative strategy over the opportunistic. For example milk yield of the conservative strategy is 414 per lactation regardless of year, but the opportunistic scenario's milk yield is only 296. Similar premia are awarded to the conservative strategy in respect of calving rate and mortality, absolute liveweight, the division between live-sale (= high price) and emergency slaughter (low price), ploughing and transport and of manure output. These premia, which are selected in some cases on the basis of experiments, in others of 'conventional wisdom', are justified by differences in average annual stocking rates between the scenarios, and are invariant over time even though in some years the conservative scenario has a higher stocking rate than either of the other two CEA scenarios, that is, the 'opportunistic' (in essence what communal farmers do at the present) or the 'tight-tracking' (which matches what I have described as 'opportunistic' at the start of this paper).

The mimic model allows for the effects of natural reproduction and mortality, as well as sales and purchases, as methods by which stock numbers are determined. It does not (nor does the CEA original) allow for different age-sex structures of the herd. The structure of the herd, and of offtake from it, remains stable in terms of age-sex composition.

Prices are expected to vary over time as a consequence of changes in levels of output inversely related to quantity of output. In the absence of adequate data and analysis of price-quantity relations, CEA, copied by us, make use of a convenient rule. They calculate the maximum and minimum levels of output (in quantity terms), assume that maximum and minimum prices will occur at minimum and maximum output levels, guess a ratio between the minimum and maximum prices equivalent to these extreme output levels, and assume that prices change between these levels in exact proportion to output levels. Distinct price levels for each year of the simulation are then obtained through a formula relating actual prices in 1996 to model-derived quantities of output in 1996.

As can be seen the CEA-mimic model is highly simplified in both its biophysical and economic structure. On the other hand it integrates (as the bio-physical models do not), biophysical and economic factors. It is in principle generalizable over a wide range of ecologies and geographical areas and its 'convenient rule' mentioned above avoids the need for time-consuming and expensive surveys.

The Sandford model modifies the mimic model. It allows variations in the age-sex structure of the herd, and, instead of the invariant premia awarded to the conservative strategy by the mimic model, it provides for technical coefficients, (calving, mortality, milk-yield and weight) to vary with grazing pressure (expressed as cattle population in year t related to a weighted average of rainfall-induced vegetation in years t and t-1). It does this by extending the 'convenient rule' applied to the relationship between maximum and minimum quantities of output and prices in the mimic model also to the relationship between rainfall and output. For example, if 400 litres is the maximum lactation yield in the best rainfall year and 250 litres in a drought year is the worst, then a rainfall year half way in quantitative terms between the worst and the best will trigger a lactation yield of 325 litres.

Some of the 'proof of the pudding is in the eating', and I have tried to 'taste' these models by comparing one result, the relationship between stocking rate and output, between them. Since the units of measurement involved differ between studies and models I have had to do this in proportional terms rather than absolute numbers. The results are shown in Table 2.

All the models were designed for or applied to semi-arid rangelands in southern Africa with an annual average rainfall between 550 and 625. The table shows, for each study or model, the proportionate increase in output (liveweight gain, Column 3) and the associated rise in stocking rate (Column 2) at higher levels of output and stocking rate, both expressed as percentages of the lowest output/stocking rate in the study concerned. The fourth column expresses the ratio between the increase in stocking rate and the increase in liveweight gain. In the table I have divided the studies into two sorts. One, called 'bio-physical models' makes extensive use of the scientific literature to enable a model to mimic faithfully the actual biophysical processes. The other sort, labelled 'models using crude short-cutting techniques' are the CEA-mimic and Sandford models described above.

It would have been convenient if the ratios in the fourth column had had values quite close to each other, since it

would have implied that the models were equally useful as predictors of performance over a wide range of ecosystems within the semi-arid rangelands,[6] and one would have had fewer qualms about using the short-cut methods which require less data and cover a wider range of outputs. As it is the bio-physical models differ quite as much between each other as they do with the short-cut methods. One is reluctantly forced to concede that, if the bio-physical models are more likely to be correct, then the short-cut models, at any rate in their biophysical aspects, do not match them closely enough. Equally one is forced to concede that the bio-physical models differ widely also, and that a large number will be required to match each of the ecosystems within each country's rangelands. It seems likely that most African states with pastoral areas will lack the resources in the next ten years to make policy-making using such models a viable exercise.

Table 2: A comparison of experiment-based results with short-cutting techniques

SOURCE-STUDY OR MODEL – **SITE** – *OPTION OR VARIANT*	PERCENTAGE INCREASE IN STOCKING RATE (LW/HA)	PERCENTAGE INCREASE IN LW GAIN (KG/HA/YR)	RATIO OF STOCKING RATE/LW GAIN INCREASES	STOCKING RATE (APPROX.) AT INCREASED LEVEL (KGS LW/HA)
BIO-PHYSICAL MODELS				
Fynn and O'Connor (2000) – **Llanwarne, Zululand bushveld, SA** – *Medium s.r.*	53	44	1.18	107
Fynn and O'Connor (2000) – **Dordrecht Zululand, SA bushveld** – *Medium s.r.*	27	11	2.37	94
Fynn and O'Connor (2000) – **Llanwarne, Zululand bushveld, SA** – *High s.r.*	101	77	1.31	141
Fynn and O'Connor (2000) – **Dordrecht, Zululand bushveld, SA** – *High s.r.*	70	40	1.74	125
Illius, Derry and Gordon (1998) – **SW Zimbabwe savanna** – *tracking-basic/ tracking-capped, with restocking and female sales*	80	32	2.50	78
MODELS USING CRUDE SHORT-CUTTING TECHNIQUES				
CEA-mimic – **Chivi, Zimbabwe, semi-arid farming** – *Tight-tracking scenario using original assumptons of CEA*	49	23	2.14	82 *
CEA-mimic – **Chivi, Zimbabwe, semi-arid farming** – *tight tracking scenario; original assumptions as modified by Sandford & Scoones*	49	40	1.22	82 *
Sandford – **Chivi, Zimbabwe, semi-arid farming** – TT-ATI / CON-ATI (NIBR version)	49	40	1.22	82 *
UNWEIGHTED AVERAGE	**61**	**38**	**1.60**	**109**

Notes to Table 2

SA = South Africa

s.r. = stocking rate

LW = Liveweight

TT=Tight tracking

CON = Conservative

ATI = Version of model where animal traction is important

NIBR = Version of model where emphasis is not given to breeding stock

* This information is not in the original source but has been gleaned, with possible error, from several sources.

HOW IMPORTANT IS THE ISSUE OF CHOICE BETWEEN OPPORTUNISM AND CONSERVATISM?

Professionals of all disciplines concerned with the management of rangelands have long been interested in the impact of stocking rate on rangelands performance. Their concern has been with both the effects of stocking rates on productivity and on rangelands degradation, although some of them may have had their concern about degradation lulled by disequilibrium theory, by studies, for example, those of Biot (1993) and Abel (1993), showing that degradation sometimes takes place at a very low rate, and by economists' use of discounting techniques which give more emphasis to what happens soon than to what happens later.

We can examine what light the simulation and other studies already described in this paper throw on the relative importance, in terms of system productivity, of stocking rate and other factors. This is done in Table 3.[7]

In Table 3 we analyse the differences in the NPVs of various runs of the CEA-mimic model. Each of these runs used slightly different combinations of alternative assumptions. These pairs of alternates were:

Type of alternative assumption	Alternate 1	Alternate 2
Discount rate	0%	8%
Degree of price variability, expressed in terms of ratio between maximum and minimum prices in years of minimum and maximum net offtake	Normal = 2.5	Price support = 1.5
Technical and related assumptions	CEA's selection	Sandford and Scoones' selection
Stocking strategy	Opportunistic	Conservative

The analysis shows the difference between, on the one hand, the sum of the NPVs of all the different combinations that contain a particular alternate and, on the other hand, the sums of the NPVs of all the combinations containing its rival alternate. Where the difference between the sums of NPVs of one pair of rival alternates is large compared to the difference arising between the NPVs of other pairs of rival alternates it can be concluded that the importance of that pair of alternatives is large.

As can be seen in Table 3, the importance of differences between stocking strategies is relatively large.

Table 3: The association between changes in NPV and alternate assumptions and controls (figures are in Zimbabwe $ and are derived from the CEA-mimic model's analysis).

Difference between conservative and opportunistic strategies	111 714 814
Difference between alternatives in price variability	70 280 071
Difference between assumptions about technical coefficients, etc.	90 386 292
Difference between discount rates	314 246 956

CONCLUSIONS

There is no escaping the importance of the economic aspects of the debate between conservative and opportunistic stocking strategies. Most of the world's pastoralists are in no position to adopt long-term environmental objectives at the cost of death from starvation in the short term. Nor can many countries as a whole afford strategies that unduly depress their long-term accumulation of the resources needed to achieve many of their other objectives. A proper balance between all objectives is needed, and economic analysis is one essential tool in assessing whether a country is getting its balance right.

The choice of the particular economic criterion by which one assesses the economic performance of the strategies is important in the choice between conservatism and opportunism. This is because different criteria assign different relative importance to the present and the future, and the timing of costs and returns is very different between the two alternate strategies. The criteria currently being used in most analyses may not reflect the economic priorities of pastoralists.

Analyses which concentrate on only one commodity are of very limited use in assessing performance in rangeland management because all African livestock systems have at least two outputs, and most of them many more than two.

Because rainfall is such an important driving variable in rangeland systems, and is so variable, one has no alternative to using simulation models in economic analyses of these systems. Otherwise one simply does not pick up either the full range of variability or the delayed feed-back that comes from the natural processes of reproduction in the herds.

Most of the simulation models that have been developed so far are very data-intensive and very specific to particular ecosystems. These have a strong bio-physical base and their economic aspects are less well-developed. Two more economically-oriented models try to cut corners in both their bio-physical and economic aspects. However within southern Africa, in a fairly similar

rainfall regime (about 600mm average per year), all these models produce results which differ widely. This gives no cause for confidence in the early usefulness of either detailed bio-physical or short-cut models in any general statement about the economic merits of conservatism and opportunism. Yet the evidence is that the choice between conservatism and opportunism has important economic as well as environmental consequences.

These conclusions suggest that protagonists of both conservatism and opportunism should become more wary about basing their claims on economics.

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[1] The author, together with I Scoones, has written a response (Sandford & Scoones 2003) to the CEA article rebutting its conclusions. That response has been submitted for publication. This is not the place to deal in detail with the disagreements between us and CEA.

[2] This may have been partially corrected by Chawatama et al. (2003, Parts I and II) which were discovered by me too late to be incorporated in the discussions or conclusions of this paper.

[3] Campbell et al. (2000) declined to make the software of their spreadsheet model available for us to experiment with; so we (Sandford & Scoones 2003) had to mimic it as far as possible on the basis of the information available in their published paper. This was incomplete so we have had to guess some of the values and relationships of which details were not given in Campbell et al's paper.

[4] For the existence of this in southern Africa see Tyson (1979:51) and Tyson & Preston Whyte (2000:321–2).

[5] In the mimic model work was started on both conservative scenarios, but was subsequently abandoned on the 'conservative-tracking' scenario because it had more 'opportunistic' than 'conservative' characteristics.

[6] Since the underlying relationship between dry-matter intake and forage availability is asymptotic and strong enough to colour all the other relationships (Richardson, Hahn & Schoeman, 2000:277), exact correspondence is not to be expected (unless exactly the same portions of the curve are covered in each place, which they are not).

[7] What follows is very amateur and will need a statistician's advice and subsequent revision or exclusion. There are problems of covariance that have not been sorted out.

REFERENCES

Abel, N. 1993. Reducing cattle numbers on southern African communal range: Is it worth it? *Range ecology at disequilibrium: New models of natural variability and pastoral adaptation in African savannas*, edited by R Behnke, I Scoones & C Kerven. London: Overseas Development Institute.

Barrett, J. 1992. *The economic role of cattle in communal farming systems in Zimbabwe.* London: Overseas Development Institute. (Pastoral Development Network paper; no. 32b.)

Batabyal, A, Biswas, B & Godfrey, E. 2001. On the choice between the stocking rate and time in range management. *Environmental and Resource Economics*, 20(3):211.

Behnke, R, Scoones, I & Kerven, C (eds). 1993. *Range ecology at disequilibrium*. London: Overseas Development Institute.

Belli, P, Anderson, J, Barnum, H, Dixon, J & Jee-Peng Tan. 1997. *Handbook on economic analysis of investment operations.* Washington, DC: Operations Policy Department, Learning and Leadership Center, World Bank.

Biot, Y. 1993. How long can high stocking densities be sustained? In *Range ecology at disequilibrium: New models of natural variability and pastoral adaptation in African savannas*, edited by R Behnke, I Scoones & C Kerven. London: Overseas Development Institute.

Borgerhoff Mulder, M & Sellen, D. 1994. Pastoralist decision-making: a behavioural ecological perspective, in *African pastoralist systems: An integrated approach*, edited by E Fratkin, K Galvin & E Roth. Boulder, Colorado/ London: Lynne Rienner.

Brown, L. 1971. The biology of pastoral man as a factor in conservation. *Biological Conservation*, 3(2):93–100.

Campbell, B, Doré, D, Luckert, M, Mukamiri, B & Gambiza, J. 2000. Economic comparisons of livestock production systems in communal grazing lands in Zimbabwe. *Ecological Economics*, 33:413–38.

Carande, V, Bartlett, E & Gutierrez, P. 1995. Optimization of rangeland management strategies under rainfall and price risks. *Journal of Range Management*, 48(1).

Chawatama, S, Ndlovu, L, Richardson, F, Mhlanga, F & Dzama, K. 2003. A simulation model of draught animal power in smallholder farming systems. Part I: Context and structural overview. *Agricultural Systems*. 76(2):415–40.

Chawatama, S, Ndlovu, L, Richardson, F, Mhlanga, F & Dzama, K. 2003. A simulation model of draught animal power in smallholder farming systems. Part II: Model evaluation and application. *Agricultural Systems*. 76(2):441–5.

Coppock, D. 1994. *The Borana Plateau of southern Ethiopia: Synthesis of pastoral research, development and change 1980-1991*. International Livestock Centre for Africa. Addis Ababa.

Dahl, G & Hjort, A. 1976. *Having herds: Pastoral herd growth and household economy.* Stockholm: Department of Social Anthropology, Stockholm University.

Danckwerts, J. No date. *A socio-economic study of veld management in the tribal areas of Victoria Province.* Salisbury: Department of Agriculture, University of Rhodesia.

De Ridder, N & Wagenaar, K. 1984. A comparison between the productivity of traditional livestock systems and ranching in eastern Botswana. *ILCA Newsletter*, 33:5–7.

Dorfman, R. 1962. Basic economic and technologic concepts: A general statement, in *Design of water resource systems. New techniques for relating economic objectives, engineering analysis and governmental planning*, by A Maass, M Hufshmidt, R Dorfman, H Thomas, S Marglin & G Fair. London: Macmillan.

Fratkin, E, Galvin, K & Roth, E (eds). 1994. *African pastoralist systems: An integrated approach.* Boulder, Colorado/London: Lynne Rienner.

Freer, M, Moore, A & Donnelly, J. 2002. *The GRAZPLAN animal biology model for sheep and cattle and the GrazFeed decision support tool.* (Last revision, September 2002). Collingwood, Australia: Commonwealth Scientific and Industrial Research Organisation. (CSIRO Plant Industry technical paper.)

Fynn, R & O'Connor, T. 2000. Effects of stocking rate and rainfall on rangeland dynamics and cattle performance in a semi-arid savanna, South Africa. *Journal of Applied Ecology*, 37:491–507.

Heeren, U. 1990. *The commercial sale of camel milk from pastoral herds in the Mogadishu hinterland, Somalia.* London: Overseas Development Institute. (Pastoral Development Network paper; no. 30a.)

Herskovits, M. 1926. The cattle complex in East Africa, *American Anthropologist*, 28.

Illius, A, Derry, J & Gordon I. 1998. Evaluation of strategies for tracking climatic variation in semi-arid grazing systems. *Agricultural Systems*, 57:381–98.

Maass, A, Hufshmidt, M, Dorfman, R, Thomas, H, Marglin, S & Fair, G. 1962. *Design of water resource systems. New techniques for relating economic objectives, engineering analysis and governmental planning.* London: Macmillan.

Mace, R. 1988. *A model of herd composition that maximises household viability and its potential application in the support of pastoralists under stress.* London: Overseas Development Institute. (Pastoral Development Network paper; no. 26b.)

Mace, R & Houston, A. 1989. Pastoralist strategies for survival in unpredictable environments: A model of herd composition that maximises household survival. *Agricultural Systems*, 31:185–204.

Niamir-Fuller, M & Turner, M. 1999. A review of recent literature on pastoralism and transhumance in Africa, in *Managing mobility in African rangelands*, edited by M Niamir-Fuller. London: Intermediate Technology.

Niamir-Fuller, M (ed). 1999. *Managing mobility in African rangelands.* London: Intermediate Technology.

Richardson, F, Hahn, B & Schoeman, S. 2000. Modelling nutrient utilisation by livestock grazing semi arid rangeland, in *Modelling nutrient utilization in farm animals*, edited by J McNamara, J France and D Beever. New York: CABI:263–80.

Rodriguez, A & Roath, L. 1987. A dynamic programming application for short-term grazing management decisions. *Journal of Range Management*, 40(4).

Sandford, S. 1982. Pastoral strategies and desertification: Opportunism and conservatism in dry lands, in *Desertification and development. Dryland ecology in social perspective*, edited by B Spooner and H Mann. London: Academic Press.

Sandford, S. 1983. *Management of pastoral development in the Third World.* Chichester/London: Wiley/Overseas Development Institute.

Sandford, S & Scoones, I. 2003. *Opportunistic and conservative pastoral strategies: Some economic arguments.* (Draft submitted for publication)

Scoones, I. 1990. *Livestock populations in the household economy: A case study from southern Zimbabwe.* Unpublished doctoral thesis. University of London.

Scoones, I (ed). 1994. *Living with uncertainty: New directions in pastoral development in Africa.* London: Intermediate Technology.

Snyman, H. 1999. Soil erosion and conservation, in *Veld management in South Africa* edited by N Tainton. Pietermaritzburg: University of Natal Press.

Spooner, B & Mann, H (eds). 1982. *Desertification and development. Dryland ecology in social perspective.* London: Academic Press.

Sugiharto, T & Pannell, D. 1998. A stochastic dynamic programming model for management of sheep grazing annual pastures: pasture deferment and stocking rate, in *Research profile: Agricultural and resource economics at the University of Western Australia in 1998,* edited by R Fraser. Perth: University of Western Australia:175–88.

Tainton, N (ed). 1999. *Veld management in South Africa.* Pietermaritzburg: University of Natal Press.

Tyson, P. 1979. Southern African rainfall: Past, present and future, in *Proceedings of the symposium on drought in Botswana: National Museum Gaborone, Botswana (5–8 June 1978)*, edited by M Hinchey. Gaborone: Botswana Society and Clark University Press.

Tyson, P & Preston Whyte, R. 2000. *The weather and climate of southern Africa.* 2nd ed. Cape Town: Oxford University Press.

WHY IS IT SO DIFFICULT TO TRANSLATE RANGELAND NON-EQUILIBRIUM THEORY INTO PASTORAL DEVELOPMENT PRACTICE?

Wolfgang Bayer and Ann Waters-Bayer

INTRODUCTION

Ten years after Woburn II, we are still convinced that the non-equilibrium theory offers a better explanation of rangeland dynamics in arid areas than does the conventional theory of succession, yet general acceptance of a flexible, adaptive approach to range management is restricted to a relatively small group of 'believers'. What is the problem? Lack of data? Hardly. The more data become available from long-term experiments and range monitoring, the firmer becomes the scientific basis for the non-equilibrium argument in the drylands. Is it simply a matter of slow paradigm shift, facing the common phenomenon of 'the empire fights back', or have we been overlooking other aspects that determine whether theory is translated into practice?

A first sign that it wouldn't be easy to translate the non-equilibrium theory into practice came early. Despite strong evidence of 'non-equilibrium rangelands' and despite publication on cycles of expansion and contraction of the Sahara desert in *Science* (Tucker et al. 1991), the United Nations Environmental Programme (UNEP) went ahead to warn of an alarming expansion of dryland degradation – putting the blame primarily on grazing animals (UNEP 1992). And, of course, the UN – indeed, a special agency set up to combat desertification – was prepared to come to the rescue, and issued a call for immediate action and corresponding funds.

When Camilla Toulmin took part in the negotiations on the international treaty to combat desertification in the mid-1990s, she probably managed to convince some specialists, but after her return she admitted: 'We lost' (pers. comm. 1995). Her arguments, based in part on the insights about non-equilibrium environments, were not enough. Even if there had been more evidence published in double-refereed scientific journals, would this have changed the power relations supporting the desertification theory?

POWER DETERMINES 'TRUTH'

In his article 'From theodolite to satellite', Adrian Mackenzie (2000) makes an important point: orderly people are deeply disturbed by the apparent chaos in the rangelands. Range monitoring – like other survey work – is carried out to discover the order of things, to indicate possibilities of how to control these ecosystems and to put them to stable, predictable long-term use. It is not only a matter of seeking order and certainty in an innately uncertain environment; it is also a question of power. Bureaucrats, rangeland advisers and administrators, scientists, tourist operators, government ministers, pastoralists, agro-pastoralists, smugglers, miners and many other groups have strong but very particular interests to use the rangelands to their advantage. And those who have power are unlikely to give it up easily. It is the influence that one can exert to the benefit of oneself or one's group that determines which 'scientific truth' is listened to.

This is not to say that pastoralists in a position of power pursue the wisest policies regarding rangelands. Take the example of Australia. Here, the government decided in the early 1990s to abolish drought subsidies. It was recognised that, in semi-arid and arid regions, drought is a fact of life. Until then, in some districts, 38 of the previous 40 years had been classified as 'drought' years, entitling the pastoralists to government support. It was not only the evidence of non-equilibrium environments that led to the turnaround, but also the loss of importance and therefore of power of the pastoralists. Australian pastoralists had once been the 'kings in grass castles', with strong political influence. But, by the early 1990s, because of the crises in the wool and meat market, the growth of industry and services, and the growing importance of more intensive cultivation, rangeland-based livestock production (70% of the country is classified as rangelands) was contributing only 3% to Gross Agricultural Product and the totality of agriculture

was contributing only 3% to the Gross National Product. When faced with budget constraints, the Australian government found itself in a strong enough position to reject the pastoral lobby and to abolish the drought subsidies. It was only at this time that the paradigm of non-equilibrium rangelands became interesting for supporting government policy decisions.

NEW PARADIGM DISTURBS USEFUL CONSTRUCT OF REALITY

In other countries, what reasons do the people in power have to want to accept the non-equilibrium paradigm? Let's take an example of an undisclosed country in Africa, where a consultancy mission was carried out to look into range degradation. The complaints were: diminishing rainfall, overstocking and unproductive pastoral herds. The government officials used these arguments to justify the establishment of a government ranch. When rainfall records were compiled, the apparent trend was an increase in rainfall over the last nine years for which data were available. When the grazing areas were inspected, there was no clear sign of degradation. Furthermore, fire removed much more vegetative material than did grazing. A more detailed investigation – based on interviews with herders about the life histories of the animals – revealed that the government ranch was not bad in terms of animal production (comparable to extensive ranching enterprises in Australia and the USA), but the difference in fertility and weaning percentage between large traditional pastoral herds and the government ranch was much smaller than the government officials had expected. When official statistics were consulted, these suggested that livestock numbers were decreasing in the area.

Government officials confronted with these findings started to question the reliability of their own statistics, rather than question their assumption of range degradation and overstocking. An ecological crisis, however fictive, was needed to justify development projects, which – in turn – provided income and justification for a government department and services to support 'modernisation' of livestock keeping. The new paradigm disturbs this useful construct of reality and therefore needs to be warded off. (A closer look into the behaviour of some international organisations may lead to similar conclusions.) Admittedly, it is difficult to operate in an effective way if 95% or more of the budget is for personnel. You need extra funding if you want to do more than just sit in a slowly decaying office, and you need some justification to attract funds.

PROJECT FOLKLORE AND PROJECT LOGIC

During a consultancy in a Sahelian country, a somewhat different scenario was observed. Here, the project personnel told the story that, during the 1984 drought, an eminent ecologist had declared the pastoral zone as 'finished'. When good rains occurred the following year, the pastures started greening up again and – so the story goes – the eminent ecologist was so embarrassed that he never came back to that country. Thus, some evidence of the non-equilibrium phenomenon has found its way into project folklore. However it has not led to a change in project approach. The project helped set up an early warning system, with geographic information systems and interpretation and calibration of satellite imagery, producing beautiful maps. But the logical accompanying step – drawing up a drought contingency plan – was not taken. Furthermore, there was no sign that the early warnings were actually reaching the pastoralists. Project activities were geared towards subsidising food, providing food-for-work or cash-for-work and introducing holistic resource management, based on the idea that rotational grazing would produce more grass than under traditional pasture management.

From a project point of view, an argument against drought contingency planning might be that little can be shown if no drought occurs, and both pastoralists and donors evaluate projects on the basis of what they can show. Project cycles are short – often no more than three years. What happens if the three years happen to be good? The value of drought contingency planning cannot be proved. Despite support from the World Bank and other donors (de Haan et al. 1997), not all donors are prepared to buy the non-equilibrium theory and, even if the donor organisation may support studies such as the one by de Haan et al. not all desk officers within the organisation are convinced. Furthermore, some projects may be initiated with the purpose of 'bribing' local people to support the government of the day. Development efforts based on the non-equilibrium paradigm – which require organisational and institutional innovation rather than huge and expensive infrastructure – are poorly suited for this purpose.

STORYLINE TO SUPPORT CONVENTIONAL RESEARCH

These examples have referred to policy and planning of pastoral development projects in African countries. But we can also cite examples from research outside of Africa. One of us is a member of the German Association of Desert Ecology, which – for lack of deserts in Germany – focuses on the South. During the annual meetings in Germany, botanists and geographers regularly present data (normally short-term – a PhD student cannot do fieldwork for five years or longer) on vegetation, stressing how fragile the ecological balance is in arid areas (which seems to contradict the fact that flora and fauna manage to survive sometimes several years without rain) and trying to prove degradation. Presentations on the non-equilibrium nature of vegetation in arid areas are politely listened to, but the contradiction to other presentations is overlooked with similar politeness. We suspect here that proclaiming

range degradation is a way of securing funding for research programmes. Moreover, it is difficult for some scientists to accept new ideas if they are not their own. But basically it comes back to the same thing: it is not scientific evidence or even common sense that drives the agenda in research. As in development; it is the storyline that is spun to support the interests of those in power.

REALITY MORE COMPLEX THAN THEORY

There is yet another complication: as neat and tidy as the non-equilibrium theory initially seems to be, those few policymakers who are really prepared to consider it may recognise that reality is not so clear-cut as the theory seems to suggest. Equilibrium and non-equilibrium paradigms are only models to explain the behaviour of vegetation. Real vegetation is rarely purely equilibrium or purely non-equilibrium; it is somewhere in between. Trees or shrubs may even exhibit both equilibrium characteristics (for example, if they are cut or browsed) and non-equilibrium characteristics (during seedling establishment). In some cases, such as *Artemisia herba-alba*, the plants depend on the 'right' grazing pressure, however defined: undergrazing leads to woody growth and death of the plants within a few years, while overgrazing does not kill the plants, but they remain small and grow only very slowly. In practice, however, much cruder measures than grazing pressure affect these plants: ploughs in the steppes disturb the vegetation so profoundly that it can take decades until the pre-ploughing vegetation can re-establish itself. Other crude tools include axes and pickaxes to cut down trees and shrubs and even to dig up roots to obtain wood for charcoal making. In these arid areas, it is not overgrazing but rather ploughs and axes that are degrading rangeland vegetation.

EMPOWERING PASTORALISTS TO CHANGE THE STORYLINE

If non-equilibrium theory is to be seriously taken into account in policymaking, then the people who will benefit from getting this story across have to gain a stronger voice in influencing policy, in making the alternative storyline more loudly heard. Currently, the stakeholders in degradation still have the upper hand. If it suits them, because of scarcity of funds and weak pastoral influence, governments and donors may even find the non-equilibrium paradigm useful to justify not giving any attention to pastoral development. The only way to prevent this happening is to strengthen the lobbying power of pastoralists and this means strengthening pastoral organisation.

In another country with large pastoral areas – Ethiopia – a Pastoral Community Development Project (PCDP) was recently planned, supported by the World Bank and the International Fund for Agricultural Development (IFAD) (World Bank 2003). The PCDP is built on concepts of the non-equilibrium theory and promotes a mobile and flexible form of rangeland use with decentralised decision-making. The project planning team was up against government policy that still stressed the need to settle nomads and to increase irrigated crop production in river valleys. These are often vital seasonal grazing grounds and important sources of water for livestock. The government officials genuinely want to provide services, but grapple with the problem of providing formal education, human and animal health services and security, among other things, to a sparse and mobile population.

Why has the Ethiopian government decided to try an alternative approach to pastoral development – supporting mobile pastoralism – alongside its continued efforts to settle nomads? Because the World Bank and IFAD support the PCDP to the tune of $50 million. A major effort is now underway to familiarise regional and national heads of services with pastoral/non-equilibrium thinking. This involves workshops, study tours and training courses, with strong support from the UK's Department for International Development (DFID) and some Ethiopian experts in pastoral development – mainly in NGOs – and some members of parliament from pastoral areas of Ethiopia. It remains to be seen whether the arguments and evidence that will hopefully be generated in the first five years of the project will be sufficient to convince the pro-sedentarisation policymakers of the merits of this alternative approach to pastoral development. The PCDP is banking on the hope that, by strengthening pastoral organisation and decision-making power over local use of funds, the pastoral peoples will gain power to lobby for their rights to manage their resources in appropriate ways for non-equilibrium environments. It is an experiment worth watching.

REFERENCES

De Haan, C, Steinfeld, H & Blackburn, H. 1997. *Livestock and the environment: Finding a balance.* Brussels: European Commission.

Tucker, C, Dregne, H & Newcomb, W. 1991. Expansion and contraction of the Sahara Desert from 1980–1990. *Science,* 253:299.

Mackenzie, A. 2000. From theodolite to satellite: Land, technology and power in the Western Division of NSW, in *Agricultural extension and rural development: Breaking out of traditions*, edited by R Ison & D Russell. Cambridge: Cambridge University Press:80–102.

UNEP. 1992. *Status of desertification and implementation of the United Nations plan of action to combat desertification.* Nairobi: UNEP.

World Bank. 2003. Pastoral Community Development Project. http://www4.worldbank.org/sprojects/Project.asp?pid=P075915

EMERGING THEMES AND FUTURE DIRECTIONS

SUSANNE VETTER

It became apparent during the workshop, and the International Rangelands Congress (IRC) which followed it, that much of the initial heat of the debate has dissipated. Recent years have seen the emergence of an increased awareness of the spatial heterogeneity and temporal variability of semi-arid and arid rangelands. There is increased acknowledgement that rangeland management in drylands is complex and is influenced by spatial, bio-physical, social, cultural and economic factors at a multitude of temporal and spatial scales. A commonly expressed view at the workshop was that there is more to rangeland dynamics than can be explained by equilibrium and disequilibrium theory alone. Whether the debate will seek out new dichotomies (for example, key resources vs non-equilibrium resources, mobility vs fragmentation) or steer clear of them, remains to be seen. Perhaps the most important message that came out of the workshop was that it is more important to understand how and why systems differ than to search for a single model that describes all places.

Two questions central to the original debate received relatively little attention at the workshop: Are stocking rates important, and does degradation occur in non-equilibrium rangelands? Here, too, the debate has moved on from the dichotomy that led to such a heated stand-off in the mid-1990s. It is now widely acknowledged that while many assessments of degradation were exaggerated and their attributed causes have been oversimplified, degradation has occurred in many semi-arid rangelands. The causes of this are complex – and the proximate causes and underlying causative processes have to be unravelled carefully – but often result from sedentarisation of pastoralists or supplementary feeding, both of which lead to continuous, heavy utilisation of parts of the range. The effects are usually spatially heterogeneous and often difficult to quantify, especially effects on secondary production, which tend to be masked by spatial heterogeneity (Ash et al. 2002). Usually degradation takes place over timescales much greater than those at which management decisions are made, and this disparity in scales has led land users not to perceive degradation as a concern.

More and more studies suggest that the sustainability of non-equilibrium rangelands is dependent on drought (or other factors) periodically reducing stock numbers and thus keeping grazing pressure below levels that are likely to cause degradation over the long term. A similar effect is achieved by moving livestock to less drought-affected parts of the landscape. An important question is whether this post-drought reduction in grazing pressure is a general prerequisite for rangeland sustainability. There is concern about the effects of providing supplementary feed on rangeland resources, as this allows high grazing pressure to be maintained in an area during and after the dry season. The provision of large amounts of subsidised supplementary feed, as is common in north Africa, the Middle East and China and was widespread in northern Asia during the Soviet era, has been observed to result in rangeland degradation (Seligman & Perevolotsky 1994, Kerven this volume).

Spatial heterogeneity and scale dependence are now recognised as defining features of drylands. Scale dependence results from spatial and temporal heterogeneity. The quality, quantity and seasonal availability of forage differs between parts of the landscape, and some areas are also more resilient to degradation than others, either because livestock cannot access them for prolonged periods (for example, annual grasslands, grazing areas far from permanent water), or because the dominant plant species are tolerant of heavy defoliation (for example, stoloniferous grasses). When phenomena are scale dependent, inferences about large-scale behaviour cannot reliably be made on the basis of smaller-scale observations (for a recent discussion see Hobbs 2003). Many range ecologists are struggling to overcome the mismatch between the scales of ecological investigation and those at which ecological processes in rangelands take place. While there is now a plethora of

experimental results at the plot scale (thanks to decades of rangeland research), data from heterogeneous landscapes are still scarce.

Climatic variability is an important driver of semi-arid and arid pastoral systems. Traditionally, pastoralists have employed mobility, flexibility and reciprocal networks to ensure access to different rangeland areas and grazing reserves at different times. This appears to be a ubiquitous feature of pastoral systems in different parts of the world (Fernandez-Gimenez & Swift 2003). Much of this mobility has been constricted as rangelands are becoming fragmented through privatisation, fencing and the transformation of grazing land to crop cultivation and settlements. The causes and extent of fragmentation, its costs to pastoralists and the environment, and possible ways of reversing or mitigating it are presently the subject of policy debate (Niamir-Fuller 1999a & b) and large-scale research (for example, SCALE).[1] Options for buffering the effects of temporal variability are moving livestock into other areas, providing supplementary feed, selling and restocking or a combination of the above. The viability of these options in different pastoral systems, and their ecological and economic consequences need to be better explored.

Progress in the debate has been hindered by a lack of clarity on the types of systems under discussion. Nomadic pastoral systems, more settled agropastoral systems and commercial ranching are all subject to temporal variability and spatial heterogeneity, but management and policy options are different in different types of rangeland systems. The research, management and policy dimensions of the debate have narrowly concentrated on two components of the system, forage and livestock, and how they interact and affect each other. This ignores other important components of livelihoods in rangelands, such as harvesting and trade in other natural resources, crop cultivation and migration in and out of the pastoral system. Apart from differences that have evolved in traditional systems under the influence of different climatic and ecological constraints (for example, Ellis & Galvin 1994), rangelands across the globe have been affected by a variety of other factors such as population growth, encroachment of other land use on rangelands, restriction of mobility, government policies and interventions, access to healthcare and education, urbanisation, and the different aspirations of the younger generation. Various combinations of these factors have led to far-reaching and often profound changes in the livelihood strategies of pastoralists, and in many areas, livestock make a decreasing contribution to livelihoods.

Despite improved consensus or at least communication among researchers, the translation of research findings into management recommendations and policy has been very slow. Some of this has to do with the difficulty in extrapolating results from controlled experiments to larger scales. It is difficult to make confident recommendations in unpredictable systems, and many researchers appear reluctant to do so. There is also considerable resistance at the policy level to communal tenure, mobility and other flexible land use practices because those too are harder to control and predict. And while the concept of adaptive management is widely considered to be sensible and appealing, changing the laws and policies to allow and facilitate it is not easy. The time scales at which adaptive management is implemented and monitored exceed conventional research and development planning horizons (see Bayer & Waters-Bayer, this volume), and this further explains why it is so rarely put into practice.

There has been a growing recognition of the need to integrate the ecological, economic, social and institutional dimensions of rangeland research. Nevertheless, problems communicating across disciplines still persist, and there remains a tension between those who see the debate mainly in terms of ecological theory and those who see it in a larger socio-political context. Some of the latter feel frustrated at the detached approach of many ecologists and feel that policy questions should inform the (scientific) research agenda. A persistent problem in these discussions is that pastoralists are still underrepresented at defining the research agenda with their needs, priorities and knowledge. They remain in most cases subjects of research, development and policy rather than playing an active role.

BOX 1: WHAT PROGRESS HAS BEEN MADE IN UNDERSTANDING THE DYNAMICS OF RANGELANDS?

- A move away from the 'equilibrium–non-equilibrium' dichotomy towards acknowledgement of the greater complexity of rangeland dynamics.
- A better understanding of the heterogeneity and scale dependence of rangeland processes.
- Wider acknowledgement of the temporal variability and unpredictability in rangelands, and the need for flexible management strategies to address this.
- Greater recognition of the interdependence of ecological, economic, social and institutional dimensions of rangelands.
- Recognition of larger-scale drivers beyond the control of pastoralists, such as political, demographic and climate change.
- A move to more interdisciplinary research covering larger time frames and spatial scales, for example, SCALE.
- A better understanding of the relationship between livestock population dynamics and

resource heterogeneity and particularly the role of key resources (Illius & O'Connor 1999; 2000; this volume).

- A less polarised view on degradation where claims of degradation and its purported causes are more critically examined. It is acknowledged that many non-equilibrium systems have experienced degradation.[2]
- A realisation that much degradation of pastoral systems has been caused or exacerbated by fragmentation, sedentarisation and a loss of mobility, flexibility and access to alternative resources and opportunities.
- A recognition that average stocking rates are less important in determining the impact of herbivores on the vegetation than the timing, duration and distribution of grazing by different herbivore species. A measure of grazing pressure (livestock units per unit of available forage) is more informative about the impact of grazing than stocking rate per se.
- Empirical studies investigating the interaction between grazing and rainfall effects on vegetation have illustrated that both have an influence over a large range of climatic conditions but their relative importance varies (for example, Hiernaux, this volume; Fernandez-Gimenez & Allen-Diaz 1999).
- The scope of enquiry into drylands and pastoralism has expanded to include areas outside Africa, and this has allowed the examination of similarities and differences in ecological dynamics and pastoral strategies between different systems (for example, Fernandez-Gimenez & Swift 2003; Kerven, this volume).
- All places are not the same – it is more important to describe why places are different than finding a model that fits all places.

BOX 2: RESEARCH AND POLICY CHALLENGES IN THE MANAGEMENT OF RANGELANDS

- There is a need to test the prediction that the risk of degradation of non-key resources increases as the ratio of key to non-equilibrium resources increases. In focusing on livestock population dynamics, does the key resource concept distract from the impact herbivores have on the vegetation?
- What is the nature of key resources in different systems, and is there a clear distinction between key resource and non-equilibrium resource as the Illius and O'Connor model suggests?
- Overcome the mismatch in temporal and spatial scales of ecological investigation and ecological processes in rangelands (multiple influencing variables, heterogeneity, especially in processes such as degradation).
- It is becoming clear that the susceptibility to degradation of different areas and patches within areas differs. What determines vulnerability and resilience at different scales? Can this be generalised?
- What determines post-drought resilience? Is reduction in grazing pressure after droughts necessary to sustain productivity?
- Natural destocking (mortality) causes great suffering. How can one destock and restock to avoid this?
- What are the options for buffering against climatic variability? What are their ecological and economic consequences, and which ones are viable options in different pastoral systems (for example, supplementary fodder, movement to other areas, destocking and restocking)?
- What are the costs of fragmentation and loss of mobility to pastoralists? Are there any benefits?
- How can one re-aggregate fragmented landscapes, or compensate for the negative effects of fragmentation? Under what conditions is restoration of mobility feasible, and what models can be developed that meet pastoralists' diverse needs (for example, for rearing live-stock but also accessing markets, education and healthcare)?
- How can we better predict the ecological and economic outcomes of different pastoral strategies in highly variable environments (see Richardson et al. this volume; Sandford, this volume)?
- How does one understand and integrate drivers at different scales, from localised defoliation and patch selection to globalisation and macro-economic policies? In which systems is the debate about ecological dynamics still important – and where do other factors override the ecological dynamics?

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[1] Scale and Complexity in Arid Land Ecosystems. http://www.nrel.colostate.edu/projects/scale/

[2] A more integrated paradigm that explicitly examines causal processes, effects and their interactions at

different scales has recently emerged (the Dahlem Desertification Paradigm (DDP), Reynolds and Stafford Smith 2002; 2003).

REFERENCES

Ash, A, Stafford Smith, D & Abel, N. 2002. Land degradation and secondary production in semi-arid and arid grazing systems: What is the evidence? In *Global desertification: Do humans cause deserts?* Edited by J Reynolds & D Stafford Smith. Berlin: Dahlem University Press.

Ellis, J & Galvin, K. 1994. Climate patterns and land-use practices in the dry zones of Africa. *BioScience*, 44:340–9.

Fernandez-Gimenez, M & Swift, D. 2003. Strategies for sustainable grazing management in the developing world, in *Proceedings of the VIIth International Rangelands Congress, Durban, South Africa.*

Hobbs, N. 2003. Challenges and opportunities in integrating ecological knowledge across scales. *Forest Ecology and Management*, 181:223–38.

Niamir-Fuller, M. 1999a. International aid for rangeland development: Trends and challenges, in *People and rangelands: Building the future: Proceedings of the VI International Rangelands Congress, Townsville, Australia* vol 1, edited by D Eldridge & D Freudenberger:147–53.

Niamir-Fuller, M. 1999b (ed). *Managing mobility in African rangelands: The legitimization of transhumance.* London: Intermediate Technology.

Illius, A & O'Connor, T. 1999. On the relevance of nonequilibrium concepts to semi-arid grazing systems. *Ecological Applications*, 9:798–813.

Illius, A & O'Connor, T. 2000. Resource heterogeneity and ungulate population dynamics. *Oikos*, 89:283–94.

Seligman, N & Perevolotsky, A. 1994. Has intensive grazing by domestic livestock degraded the Old World Mediterranean grasslands? In *Plant-animal interactions in Mediterranean-type ecosystems*, edited by M Arianoutsou & R Groves. Dordrecht: Kluwer.

The debate on equilibrium vs non-equilibrium dynamics in pastoral systems emerged in the 1980s and gained momentum in the early 1990s after two international workshops around emergent new paradigms in rangeland ecology and socio-economics (Woburn I and II).

This volume comprises nine papers presented at a workshop held at the VIIth International Rangelands Congress in Durban, South Africa, in July 2003. The aim of the workshop was to bring together an inter-disciplinary group of researchers and practitioners to take stock, ten years after the Woburn workshops, of where the debate has moved, what research has been done to test the alternative models, and to identify directions for a future research agenda.

A commonly held view at the workshop was that there is more to rangeland dynamics than can be explained by equilibrium and non-equilibrium theory alone.

These papers illustrate the diversity of approaches and some of the challenges encountered in the dialogue between the disciplines with their different languages, research approaches and ideologies.

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